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BIOLOGICAL CONCEPTS FOR DESIGN AND OPERATION OF THE ACTIVATED SLUDGE PROCESS

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16. Abstract

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BIOLOGICAL CONCEPTS FOR DESIGN AND OPERATION OF THE ACTIVATED SLUDGE PROCESS

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ABSTRACT

Over the past decade, many of the thousands of experiments accomplished in the principal investigators' laboratories were designed to permit interrelated analyses and correlation of the 10-year research effort into a general body of conceptual principles applicable to design and operation of activated sludge processes. The aim of this report is to present these biological concepts in terms readily understood by engineering professionals. The document is by no means a design and operational manual, although some recommendations for design and operational procedures are included.

Generalized concepts of BOD exertion, the use of ΔCOD as a design and operational tool, the stoichiometry and mass balance concepts of treatment, and kinetic equations for microbial growth are presented. Design models are discussed, and a model for completely mixed reactors holding recycle solids, X_R, constant is recommended. Some guidelines for accommodation of various types of shock loadings are included. Concepts of oxidative assimilation and the multiple effects of solids concentration, nitrogen concentration, and detention time are related; a new activated sludge process (continuous oxidative assimilation) for nitrogen-deficient wastes is presented. Data supporting the concept of total oxidation are presented, and a modification of the extended aeration process incorporating chemical hydrolysis of portions of the sludge is recommended. In the final chapter, some possible flow diagrams for complete aerobic treatment (purification and sludge disposal) of metabolizable organic wastes are presented.

The report was submitted in fulfillment of project 17090 FQJ under the sponsorship of the Environmental Protection Agency.

Key words: activated sludge, kinetic models, shock loads, nitrogen-deficient wastes, ultimate disposal, oxygen demand, oxidative assimilation, endogenous phase, purification efficiency.

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CONCLUSIONS

The general conclusion which emerges from the decade of research investigations briefly summarized in this report is that activated sludge processes must be designed and operated with greater consideration of the biological nature of the process. Concepts based on microbial physiology can be adapted to systems employing heterogeneous populations, and application of biological principles can lead to innovative improvements in the basic process. Design of treatment processes must include provision for operational controls which will ensure reliability of performance, and operation must be based on daily assessment of purification efficiency. Both design planning and operational procedures should attempt to minimize disruptions of the process. Each treatment facility should be designed, not according to a standard flow sheet, but as a combination of unit processes chosen to accomplish specific objectives in purifying a specific waste. The activated sludge process can and should deliver reliably a high degree of purification of organic wastes, but this requires both good design and careful, intelligent operation.

Some of the specific conclusions discussed in the body of the report are listed below with reference to the pertinent chapter.

1. Insofar as design and operation of activated sludge processes are concerned, the BOD test should no longer be used as the parameter of pollutional potential or as a measure of purification efficiency. For design and for daily assessment of operation, the ΔCOD measurement should replace the BOD test (Chapter II).

2. Treatability studies should precede design and a stoichiometric balance based on measurements of ΔCOD, oxygen uptake, and ΔCOD of the biological solids is recommended for use in these studies (Chapter III).

3. Kinetic equations for microbial growth may be employed with heterogeneous populations, but no precise values for the biological constants can be expected. The values of these constants should be determined in treatability studies. In general, the following ranges seem reasonable: μ_{max}, 0.4 to 0.6 hr^{-1}; K_s, 50 to 125 mg/l; Y, 0.4 to 0.6 (Chapter IV).

4. A kinetic model was developed specifically for use in designing activated sludge processes. The model employs recycle solids concentration, X_R, as a system constant, and it is designed to maximize purification efficiency by maintaining a "steady state" as nearly as is possible with a heterogeneous microbial population and with a waste which may vary quantitatively and qualitatively. The model requires use of an aerated sludge consistency tank and, for wastes subject to considerable variability, a surge basin is recommended (Chapter V).

5. Allowable limits for several types of shocks commonly encountered

in waste treatment were suggested. In general, it was recommended that for aeration periods of 6 to 8 hours, it should be attempted through both design and operation to prevent fluctuations in flow rate of greater than 100% of the design flow, fluctuations in concentration of metabolizable organic material in the waste of greater than 100%, fluctuations in pH of greater than one unit and rapid changes in temperature of greater than 10°C (Chapter VI).

6. The standard level of nitrogen supplementation based on a BOD:N ratio of 20:1 should be reconsidered in design of treatment facilities for wastes deficient in nitrogen. To avoid leakage of nitrogen in the effluent, treatability studies should determine the optimum combination of detention time, purification efficiency, nitrogen supplementation and nitrogen concentration in the effluent, since all of these factors are related. For treatment of wastes containing little or no nitrogen, a new modification of the activated sludge process was developed, and it is recommended that these wastes be treated by continuous oxidative assimilation (Chapter VII).

7. The extended aeration process was shown to be theoretically sound and a modification designed to allow engineering control of solids levels by chemical hydrolysis of a portion of the sludge was developed (Chapter VIII).

8. Two possible flow sheets for the activated sludge process, incorporating the kinetic model discussed in Chapter V and the hydrolytic assist to total oxidation described in Chapter VIII, were presented. These flow sheets summarize the biological principles and research findings presented in the report (Chapter IX).

RECOMMENDATIONS

One of the inevitable recommendations resulting from the past year of data analysis and the past decade of intensive research on activated sludge processes is that those designing activated sludge processes should pay greater attention to the biochemical and biological principles responsible for the behavior, i.e., the success or the failure, of the process. It has become apparent to us, and to some other researchers as well, that the potential for biological processing of waste organic substrates has barely been tapped, and that such processes will, as time goes by, become increasingly important for pollution control and possibly in the future for direct recycle of organic materials. Also, in the ultimate sense, the rivers, reservoirs, lakes, and oceans which environmental engineers and scientists must learn to control are indeed biological reactors. Thus, no matter what alternatives to biological treatment we may seek for the immediate future (and there are some depending upon the nature of the organic matter in a waste water) we eventually must face up to the need for biological engineering since the environment of which we may claim to be custodians is a biological one. And thus, as the reader scrutinizes this report and the principles, process modifications and approaches to design outlined herein, he may also employ the report in conjunction with, or use it as a springboard to, the basic microbial and biochemical literature. It is also recommended that the report be used as an aid in assessing the research literature in the environmental field.

Concerning the technological biochemical concepts and process modifications presented in the report and briefly stated in the preceding conclusions, it is recommended that the next step include larger scale developmental research relative to the design model herein suggested (Chapter V), as well as pilot plant research or field-scale testing of the continuous oxidative assimilation process (Chapter VII). Also, developmental research on the hydrolytically assisted extended aeration process (Chapter VIII), and pilot plant research on the total process flow sheets (Chapter IX) would seem a fruitful extension of the research. Also it is felt that the continued analysis of the experimental data from which tentative rough guideline limits for various shock loadings were suggested (Chapter VI) can considerably refine these guidelines.

In addition to the developmental research on principles and processes recommended in this report and further conceptual analysis of the shock load investigations we have already accomplished, there is, in general, a great need for definitive, fundamental biological research on the biomechanics of the bio-mass, i.e., heterogeneous or natural microbial populations, and on the ecology of heterogeneous microbial systems. Paradoxically, these are areas which have been shunned by most basic biological scientists as being too applied, and by most engineers as being too basic and beyond the scope of the engineer's background knowledge. Clearly, if we are to make successful inroads toward technological control and protection of the environment, this situation must be corrected. The recommendation to correct the situation by performing the needed research is made to those readers who have a proclaimed professional dedication to pollution control, particularly to those now beginning their training for this field and who will be active in it well into the 21st century.

I

INTRODUCTION

The Perspective

The activated sludge process has been increasingly employed for secondary treatment of municipal and industrial wastes containing significant amounts of organic material. The process has many "modifications," and various flow diagrams and hydraulic regimes (e.g., completely mixed as compared to plug flow), but its chief distinguishing characteristic is that the heterogeneous microbial populations (biomass) of which the sludge is largely composed, are grown in fluidized culture with provision for maintenance of the organisms in suspension by turbulent mixing and for introduction into solution of sufficient oxygen for the respiratory requirements of the microorganisms. In these characteristics the process is distinct from other secondary treatment processes, e.g., fixed bed processes such as trickling filters.

As with other secondary treatment processes, the objective of the activated sludge process is the removal of organic compounds from the waste water. The term "secondary treatment" is used here simply because it is one which has a specific significance to engineers, and it implies the traditional employment of activated sludge as a "second line of defense" regarding removal of organic matter from waste waters; primary treatment signifies a first line of defense embodying removal of initially settleable organic matter in raw waste. The term "tertiary treatment," as herein employed, is used to signify a third line of defense for removal of organic matter not removed by secondary treatment. The terms "primary," "secondary," and "tertiary" are rather meaningless unless one specifies that they are being employed to describe processes for removal of biologically usable organic substrates. As employed herein, they apply only to processes for removing organic waste materials which exert a "demand" on the dissolved oxygen resource in natural waters into which waste may be discharged. Even within this specified boundary, one must impose some qualifications or exceptions as to the subject matter which will be herein discussed, since it will be readily recalled that, as it was originally introduced, the so-called extended aeration (total oxidation) process not only combined secondary treatment with the sludge disposal process, but also combined primary and secondary treatments as well. Also, it should be recognized that fluidized biological processes (as well as fixed beds) can be employed to remove inorganic pollutants such as ammonia, which can cause a drain on the dissolved oxygen resources in the receiving stream, and that the process may also remove from waste water some organic matter which does not exert significant biochemical oxygen demand; for example, cellulose particles, which are not readily metabolized, may become entrapped in the sludge, etc. At the outset it should also be recognized that the biological processes are by no means the only ones which can be considered for removal of biologically available organic compounds from waste waters. Chemical coagulation has been employed for years to remove colloidal organic material, and in sequential combination with other processes, e.g., adsorption on activated carbon, many types of soluble organic material can be removed as well. Such a combination of processes may become a useful alternative to biological treatment, especially for relatively low strength municipal waste waters.

The point to be emphasized here is that there is no one universally recommendable group of unit processes for accomplishment of removal of organic substrates. Water reclamation plants of the future will not be describable by any sort of "standard" flow diagrams such as have been used in the past, but by groups of unit processes joined in such a manner as to accomplish rejuvenation of a specific waste water in accordance with the purity requirements of a specific environmental situation. Thus, while much stricter standards of effluent quality will undoubtedly evolve, the required effluent quality will be attained by less "standard" arrangements of unit processes to treat the waste waters. It is innovativeness rather than standardization which is needed, and it is innovativeness which is the essence of engineering.

While it is highly desirable to seek alternatives to such biological processes as the activated sludge process, it may be an even more fruitful endeavor to seek innovativeness within the biological process. It does seem fair to say that, in many situations, if biological treatment processes could be designed and operated so as to ensure reliable delivery of the degree of treatment for which they are "designed," much of the need for treatment beyond the secondary stage might be obviated. Some engineers, scientists, and technologists in the area of pollution control would welcome a replacement for biological treatment as a means of removing organic substrates from used waters. If the reasons for seeking such replacement involve frustration in attempts to understand the process, or the feeling that it is, perhaps, a process too complicated to control, these reasons are not at all soundly based. It is important for us to recognize that engineers will, in fact, be forced to gain a better understanding of the kinetics and mechanism of biological treatment systems because, regardless of how we decide to prepare water for re-use, or regardless of who makes that decision, it must be remembered that our natural receiving bodies are "in-stream" biological reactors. Thus, if pollution control engineers are to play a leading role in overall management of water quality control systems, they must lead in understanding the systems, since understanding is the most essential ingredient for technological leadership.

The "Technological Gap"

The first aim of researchers in any profession is to gain insight (understanding) regarding a natural process or phenomenon. In engineering practice, the ultimate aim is to employ such understanding to accomplish a practical goal, to satisfy a social need. When the distance between understanding and practice becomes large, there is said to be a technological gap, and it is said that there is currently such a gap between research and practice in the water pollution control field, especially with regard to activated sludge. Such gaps are natural; they always exist, and they are helpful. They are caused as often by the fact that practice gets out ahead of understanding as by the fact that understanding gets out ahead of practice. Most seem to attribute the cause (blame or credit) to the somewhat banal belief that engineers are simply appliers of bodies of known scientific information. In many regards, this is surely not always the case. For example, often because of a pressing need of society, the "appliers" must devise processes and employ them prior to the unveiling of the basic facts which permit full understanding of the processes. Such was the case with the activated sludge process. The trouble arises when the apparent success of a process engenders a lassitude with regard to seeking understanding and when theory or hypothesis (a product of the mind), often offered without the benefit of penetrating experimental proof, becomes technological dogma which is often overly simplified half-truth.

Thus, while causes for the technological gap cannot be laid on any particular doorstep, its existence is well recognized. It should be equally well recognized that researchers and practitioners have a joint responsibility to lessen (but not **to** completely close) the gap. Fortunately, its complete closure seems impossible to attain, and this is a desirable situation because it is the gap which provides the driving force for progress in the profession. The pressure for progress will come sometimes from the practicing and sometimes from the researching segment of the profession. While both groups share the responsibility for lessening the gap, perhaps the greatest weight of responsibility resides with researchers. They should not be content with presenting their results, arguments, conclusions, and recommendations for the scrutiny and criticism of other researchers in their own and allied fields, but should from time to time communicate some of the things they think they understand (and can defend with sound practical data) to the wider and ultimately more vital technological audience which includes both researchers and practitioners, the total investigational level of professional activity. It is to this aim which the present communication is dedicated.

The Nature of the Report

In the following pages we will attempt to bring into focus some major biological concepts and insights applicable to design and operation of activated sludge processes and to further understanding of biological treatment processes in general. This report is not intended to be a manual of practice for functional design and/or operation of activated sludge treatment plants. The aim is to transfer some of the concepts developed through laboratory research investigations to practicing engineers. Whether the concepts become accepted and incorporated into technological practice is, in large measure, dependent upon how well we communicate and defend them, and how well they are received by those who scrutinize them. We present them for the scrutiny of fellow professionals (both researchers and practitioners) in the interest of advancing, but surely not of writing "finis" to, development of concepts in a field of professional activity which is only now emerging. The ideas and concepts herein presented have been developed with the aid of a decade of intensive experimental activity and observation relative to activated sludge processes in our laboratories. The primary purpose for undertaking the present research project, which is culminating with the preparation of the following report, was to draw together, correlate and analyze information obtained from various individual, but related, research projects which we have conducted over the past ten years, in an effort to derive practical unifying biological concepts. We shall not attempt to cover all aspects of the work in this short report, but we shall draw upon all of this information in presenting and clarifying "Biological Concepts for Design and Operation of the Activated Sludge Process."

II

MEASUREMENT OF PURIFICATION

The Concept of Biochemical Oxygen Demand

The most widely employed measurement of effluent purification or efficiency of activated sludge processes is the amount of biochemical oxygen demanding material (BOD) which has been removed by the process. Biochemical oxygen demand is also employed in assessing the "pollutional potential" of a raw or treated waste before discharge to a receiving stream. The assets and liabilities of employment of this parameter of measurement have been argued for years by engineers, but the fact remains that the concept of BOD is valid and usable, and certainly it is used (and misused) in all segments of the water pollution control field. Today it is nearly impossible and surely unwise to neglect consideration of the idea of biochemical oxygen demand. It seems far better for pollution control professionals to clarify understanding of BOD to help ensure its enlightened application, rather than to join the chorus decrying use of the concept because of the inadequacy of the BOD test. It is necessary to make a distinction between the concept and the test. While we can defend and recommend the concept, we shall not defend the test for all of its present applications, but will recommend a more satisfactory measurement for assessing efficiency of treatment by the activated sludge process.

Through many years of expansion of population and industry, study and observation have shown that most, if not all, of the many natural and technological uses to which the water resource is put are seriously impaired or totally negated if the dissolved oxygen (DO) in surface waters is seriously depleted. It has also become widely known that the prime reason for depletion of DO is its utilization in aerobic metabolic processes by microorganisms feeding upon the organic food (i.e., carbon source or substrate) contained in municipal wastes and various industrial wastes discharged to the stream. Thus a conceptual principle has been established which recommends that the "pollutional load" should be assessed, not by measuring its amount (i.e., amount of metabolizable organic matter in the waste) but by estimating the magnitude of its effect (i.e., the amount of oxygen that will be used because of the presence of the organic matter). Such a principle is defensible in an engineering sense, because it goes directly to the heart of the matter, potential depletion of the DO resource in the receiving stream, and because it measures a colligative effect of that organic matter in a waste water which is readily available as organic carbon source to microorganisms without requiring determination of either the total amount or types of that organic matter.

On this basis, the use of the standard BOD test for estimating the "pollutional potential" of a waste, either raw or treated, which is to be discharged to a receiving stream can be defended to some extent. However, even for this application of the concept, other techniques may be preferable because of the time required for standard BOD incubation and because of inaccuracies introduced by the dilution technique itself (1).

The use of the BOD test in measuring purification efficiency cannot be defended, and it is primarily this application for which we shall recommend a different approach. It should be remembered that the purpose of biological treatment is removal of the biologically available organic matter (carbon source) and that any means of measuring the amount removed during treatment can be employed. The choice of a method for this measurement should be based upon two criteria: (1) the method used should actually measure the percentage of metabolizable (biochemical oxygen-demanding) organic matter which has been removed by treatment, (2) the method used should be sufficiently rapid so that results may be available when needed, i.e., while the opportunity exists to exert operational control over a malfunction leading to decreased purification efficiency. The standard 5-day BOD dilution bottle technique meets neither of these requirements (2).

Exertion of Biochemical Oxygen Demand

As stated above, the concept, or principle, of biochemical oxygen demand is a vitally important one and much research effort in our laboratories and elsewhere has been devoted to study of the details of BOD exertion. An examination of the sequence of events which make up the total course of BOD exertion is pertinent to a discussion of the proper application of the concept.

Figure 1 is a generalized plot based on data obtained in many experiments of various types in our laboratory. This figure shows the course of microbial growth (increase in biological solids, X) which would be observed if one placed a small inoculum (seed) of microorganisms in a vessel in the presence of a biologically available source of carbon (substrate, S) to which the microorganisms were acclimated. The "vessel" might be a BOD bottle, a batch reactor, a reach of stream, or a pond or lake. The figure also shows a plot of accumulated oxygen uptake (BOD, y), and a plot of the total amount of organic matter remaining in solution measured by a general chemical test involving the digestion of organic matter to CO_2 and H_2O in the presence of potassium dichromate under acid conditions, i.e., the chemical oxygen demand (COD) test. The total amount of substrate removed is Δ COD. This will be discussed in more detail below.

The type of data diagrammed here can be generated experimentally by plotting results of analyses of samples taken at various times and analyzed for biological solids, X (using the membrane filter technique), and for COD, S, determined on the filtrate. The course of accumulated

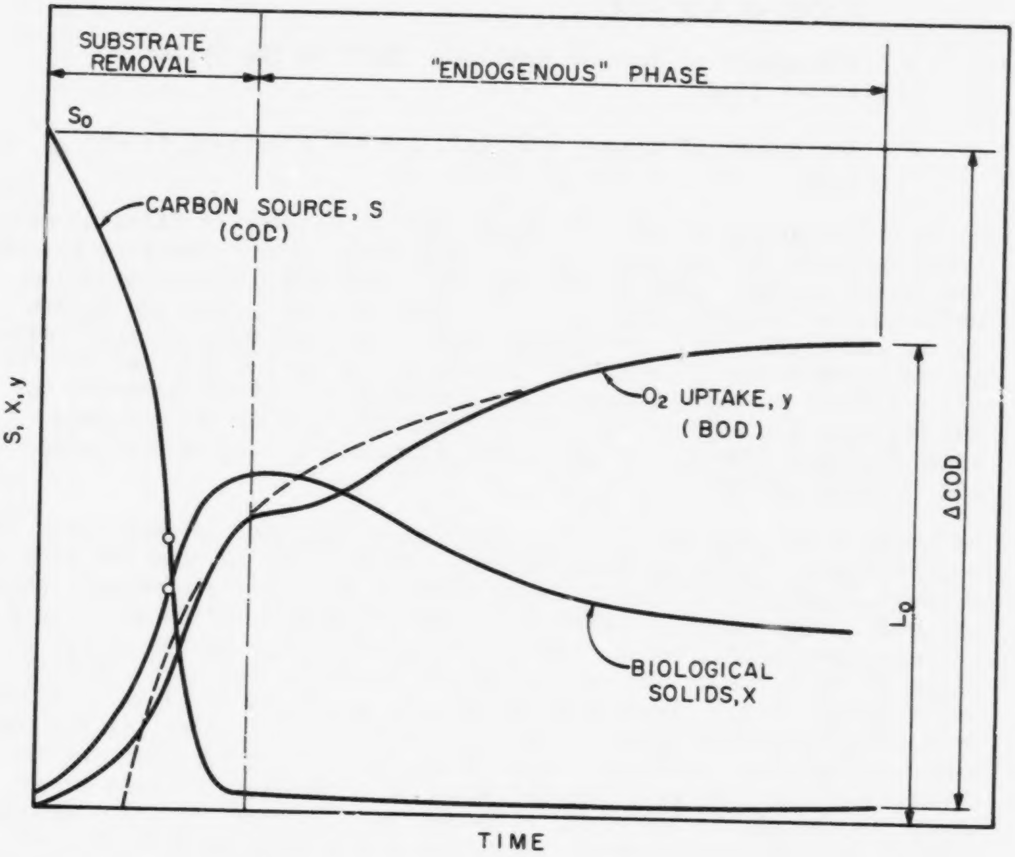


Figure 1. Generalized plot of substrate concentration, biological solids concentration, and oxygen utilization during exertion of biochemical oxygen demand. Circles mark inflection points.

O_2 uptake, BOD exertion or y, can be measured in a number of ways, e.g., by a manometric technique, or if the system is very dilute, by DO depletion in BOD bottles. Thus we have three very important parameters:

1. The amount of biological solids, X, existing in the system at any time.
2. The amount of organic substrate (COD) in the system at any time.
3. The amount of oxygen used (y), the BOD expressed at any time.

As an aid to analysis, we will assume that the system contains no nitrifying bacteria (or any other aerobic autotrophs), and therefore all of the oxygen uptake is due to the respiration of aerobic microorganisms which require organic carbon (aerobic organotrophs); thus the oxygen uptake represents only carbonaceous BOD. The experiment is terminated at the point where O_2 uptake may be continuing very slowly but has for all practical purposes stopped, and therefore all of the carbonaceous BOD has been expressed. This point along the "BOD curve" has been known for some time as the so-called ultimate BOD, and is designated by the familiar symbol, L_o.

As shown in the figure, O_2 uptake (BOD exertion) goes on long after the organic matter (COD) has been removed. It is apparent from the plot of biological solids, X, that O_2 uptake, after the substrate removal phase has ended, goes on at the expense of the biological solids which were produced during that phase. The figure also shows a decided break or pause in the curve at the termination of substrate removal. This pause or "plateau" is not always manifested; the O_2 uptake sometimes continues on a smooth ever-declining curve (see dotted lines). Also, it has been demonstrated that "plateaus" exist which do not coincide with the termination of the substrate removal phase (3). However, in most cases where the plateau is observed, it serves as a "marker" for the termination of the substrate removal phase, and its significance as such was first recognized by A. W. Busch (4). Subsequent research by Wilson and Harrison (5), and Gaudy and his co-workers (6-10) has helped to clarify understanding of BOD exertion. Much of this work has been brought together and analyzed in recent literature (2, 10, 11), and its inclusion here would add unnecessary length to this report.

Theoretical COD and BOD

In the research literature one encounters the terms "theoretical COD" and "theoretical BOD." These are useful terms, but have been subject to considerable misinterpretation and misuse. The term "theoretical COD" is synonymous with "calculated COD." It represents the stoichiometric amount of oxygen which would be required to oxidize (chemically) all of the organic matter to carbon dioxide and water. This value can be calculated if one knows which organic compound(s) are being oxidized or if one knows or can estimate the composite empirical formula for this organic matter, and if the concentrations are known. For example, if one wished to know the theoretical COD of a sample which contained 200 mg/l of sucrose (table sugar), it could be easily calculated from the following balanced equation:

$$\underset{(342)}{C_{12}H_{22}O_{11}} + \underset{(384)}{12\ O_2} \longrightarrow 12\ CO_2 + 11\ H_2O \qquad (1)$$

The theoretical COD is 200 (384/342) = 224 mg/l.

Such calculated values are useful as checks against the values obtained by the COD test, and they can be important for some research investigations. The term is not a useful one for the engineer dealing solely with whole waste problems, however, since the determination of an empirical formula for a whole waste is indeed an exercise in futility and a wasteful expenditure of investigational efforts.

The term "theoretical BOD" is sometimes employed to signify the total amount of oxygen which would be required to oxidize (biologically) to CO_2 and H_2O all of the organic substrate which is available to the organisms as food material if it were possible for organisms to do this. In accordance with this definition and with reference to Figure 1, the theoretical BOD (i.e., theoretical L_o) would be equal to ΔCOD. For this to be so would require total biological oxidation of the organic substrate in the waste. Oddly enough, some who have used this definition for theoretical BOD would also argue that it is theoretically impossible for total oxidation to occur. Total oxidation of the original organic matter in the waste would require that all of the biological solids produced during the substrate removal phase (Figure 1) were autodigested during the subsequent "endogenous" phase; i.e., the biological solids returned to the initial concentration. Such an occurrence is not a theoretical impossibility, as will be discussed in a later chapter, but the chance of its occurrence in any ecosystem in a short-term experiment (e.g., incubation of BOD's) is not very high. The degree of autodigestion of the biological solids can be expected to vary; the relative decrease in the solids level may at times be greater and at other times be less than that shown in Figure 1. The term "theoretical BOD," like "theoretical COD," is not a particularly useful one to incorporate into the practicing jargon of the field; however, it is important to know what these terms mean and what they imply when one analyzes research information (a responsibility shared equally by practitioners and researchers).

ΔCOD as a Parameter of Purification

The curves shown in Figure 1 could also be considered as representing the course of waste purification rather than BOD exertion, i.e., the emphasis may be placed upon substrate removal rather than O_2 uptake. If the waste were being purified in an activated sludge aerator, the initial solids would, of course, be much higher and the kinetics of substrate removal and solids accumulation would be affected by this difference, as will be discussed in a later chapter. However, the same reactions would be involved; carbon source would be removed, biological solids concentration would increase, and oxygen would be taken up.

In an activated sludge system of any type other than extended aeration, however, only the reactions at the left of Figure 1 are important. In the usual activated sludge aerator, the reactions occurring during the "endogenous" phase (right-hand portion of Figure 1) do not occur or, if the system approaches plug flow, may occur only briefly toward the exit end of the aerator, because the solids are separated in the clarifier. Depending upon the efficiency of flocculation and settling, the effluent contains the non-metabolized organic material (COD) which remains in the water, but little of the biochemical products of purification (biological solids). At any rate, this is the aim: retention of the sludge at the treatment plant.

Thus, at the end of the substrate removal phase the plant has accomplished its primary purpose, removal of organic carbon source which would have caused an oxygen uptake (BOD exertion) in the receiving body if this organic carbon source had not been removed from the waste. The amount of organic matter removed in the aerator is the difference between initial (influent) and terminal (effluent) COD values:

$$COD_i - COD_e = \Delta COD \qquad (2)$$

The term " ΔCOD" may be more generally defined as the amount of COD removed at any time, i.e., the difference between the COD present at the time of measurement and the COD initially present. When the amount remaining cannot be further reduced biologically, ΔCOD is a measure of the amount of organic matter in the waste sample which was available to the microorganisms. This is precisely the interpretation which many people place on BOD but, as can be seen in Figure 1, the BOD of the waste sample, L_o, is not equal to ΔCOD. It is loosely related to it, since both result from the presence of the same amount of organic matter, but the relationship can vary because the measurement of L_o requires that one consider not just removal of the carbon source, but the subsequent fate of the biomass which was produced from the organic carbon source. A significant portion of L_o is exerted after the original carbon source of the waste has served as microbial food, and the extent of this oxygen uptake is dependent upon the degree of aerobic digestion of the biological solids produced. The degree of digestion is dependent upon many factors, some of which will be discussed in a later chapter. For now it is sufficient to state that the degree of aerobic digestion will vary, and because of this, L_o can be expected to be more variable than ΔCOD.

For applications in which BOD is measured (either as BOD_5 or L_o) and is interpreted as a measure of usable organic carbon source, it seems reasonable to use a more direct, rapid, and reliable test for this parameter, i.e., ΔCOD. This recommended concept is a simple and straightforward one, but can lead to some confusion unless it is clearly delineated that we are not comparing the COD test and the BOD test. We are concerned here with ΔCOD, and the COD test is to ΔCOD what the DO test is to BOD. It should also be emphasized that insofar as testing for the amount of organic carbon source available to microorganisms is concerned, the COD remaining after the growth period (i.e., COD_e) is not really significant from the standpoint of assessing the amount of biochemical oxygen-demanding material present. The determination of ΔCOD is a measurement of only that portion of the COD of the waste which is available as biological substrate for acclimated microorganisms (or which may be otherwise removed due to the presence of the biomass, e.g., the small amount of colloidal COD which could be adsorbed on the cell surfaces, etc.). The residual COD, if sufficient aeration time has been allowed, is composed of material not utilizable by a microbial population. Thus one can employ ΔCOD to assess the amount of biochemical oxygen-demanding organic matter present, regardless of the presence of some non-biodegradable COD in the waste. For example, a waste may contain a significant amount of lignin, which is subject to chemical attack in the COD test but is not attacked to any extent biologically. Thus COD_e would be high, but would not represent a serious stream liability with regard to depletion of dissolved oxygen. It may be a detriment to the stream because of color, but not because it will be used by aerobic microorganisms at an appreciable rate. It is important to re-emphasize here that in the biological treatment process, the concern is primarily with removal of the carbon source in the waste that is used as "food" for the growth of microorganisms which are in turn removed by sedimentation (i.e., up to the time when the substrate removal phase is ended), and that ΔCOD measures this purification. On the other hand, in the BOD bottle, oxygen uptake proceeds well beyond this phase; in fact, if the curves of Figure 1 are applied to the BOD bottle, the substrate removal phase ends in one or two days. While it is a simple matter to standardize the time of incubation of BOD's, e.g., BOD_5, the actual rate of O_2 uptake is not subject to such arbitrary standardization, and use of the "standard 5-day BOD test" as a measure of plant efficiency is therefore inferior to the ΔCOD method.

Parameters for Design and Operation

The use of ΔCOD in design and operation of activated sludge processes is rather straightforward. In the experimental investigation which should precede design calculations, one can acclimate a heterogeneous population of microorganisms (e.g., from a sewage seed) by repetitive feeding cycles of the organic waste whose treatability is being investigated. During this time one can determine the extent to which the organic matter in the waste serves as microbial food (i.e., $COD_i - COD_e = \Delta COD$). Through repetitive runs a usable base level for COD_e can be established, and the fraction of the total organic matter which is amenable to biological attack can be determined. This fraction, $\Delta COD/COD_i$, can be employed as a useful parameter, efficiency of COD removal. This term is not equivalent to efficiency of purification since, depending upon the waste water under consideration, COD_e may include some organic matter which does not serve as usable food material for microorganisms. Efficiency of treatment, based upon quality of the effluent in terms of its effect on the receiving stream, would be much higher than efficiency of COD removal for such a waste.

In day-to-day operation of the process, the efficiency of COD removal can be used as a guide to assessing performance. Also, the COD_e is an important operational parameter which should be continually compared to the base line COD_e established in the treatability study which was made in the design stage. This routine check can help detect possible changes in the nature of the waste between design and operational periods and throughout the life of the plant.

The use of ΔCOD as the primary operational parameter should do much to enhance intelligent and careful daily control of the process, since information on plant efficiency is available immediately rather than 5 days later as it would be if BOD_5 is used. If COD_e remains relatively constant, there is little possibility that the actual treatment efficiency, i.e., percent removal of metabolizable organic matter, has changed significantly. However, the COD test run on the effluent does not distinguish between metabolizable and non-metabolizable organic matter remaining in the effluent, and it is therefore absolutely necessary that frequent checks on the metabolizable organic content of the effluent be made. A hypothetical example may illustrate a possible operational problem which would not be detected by use of ΔCOD as the sole criterion of efficiency. A waste containing 2000 mg/l COD, of which 1000 mg/l was not biologically metabolizable, would be completely "purified," i.e., would exhibit no further oxygen demand on discharge if COD_e were 1000 mg/l. A ΔCOD of 1000 mg/l, or a 50% efficiency of COD removal, would be established as the baseline for 100% removal of biochemical oxygen-demanding material. However, a subsequent change in the character of the waste, reducing the content of non-metabolizable material and proportionately increasing the metabolizable organics, e.g., 2000 mg/l total COD including only 500 mg/l non-metabolizable COD, might overload the plant. If the same total COD removal were continued, i.e., ΔCOD = 1000 mg/l, the COD of the effluent would remain the same but the plant would now be discharging 500 mg/l of utilizable, BOD-exerting organic matter, and the actual purification efficiency would be 1000/1500 = 67%. Such a series of coincidental alterations, yielding exactly the same results for ΔCOD and COD_e is, of course, extremely unlikely, but such a possibility emphasizes the necessity for biological testing of the effluent. For this purpose, the standard BOD test could serve to detect significant changes in purification efficiency. However, from the standpoint of rapidity of measurement and of obtaining a more credible estimate of the oxygen demand which will be exerted by the treated effluent in the receiving stream, measurement of oxygen uptake in an open, stirred jar reactor is preferable (1, 2). In addition, the plant operating engineer can and should periodically determine the ΔCOD of a sample of the waste; i.e., a sample of aeration tank influent. This can be done in an aerated batch laboratory reactor using settled activated sludge from the plant as a source of initial biological solids. Aeration of this reaction liquor should be continued until the COD of filtered samples becomes essentially constant. Comparison of the ΔCOD obtained in the laboratory experiment with the current ΔCOD through the activated sludge tank allows a direct assessment of the actual purification being achieved in the treatment plant aeration tank. A record of the values of ΔCOD and COD_e obtained in these periodic laboratory experiments and in similar experiments which were made during the treatability studies preceding design will allow detection of significant changes in the nature of the waste or in the operational efficiency of the plant.

Design engineers and plant operating engineers can work more as a team in devising changes and modifications to keep the plant "alive" if close operational supervision is initiated and maintained on a daily basis. Such a view may seem somewhat naive to some design engineers because of past and present experiences both on the design and operational sides of the coin. However, when one addresses himself to the future, it is becoming more and more apparent that design and operational technologies must merge. From the standpoint of prolonging his career, the rapidity with which pollution control plants are being designed today should be enough proof to convince any consulting engineer that his consulting capacity should include operational considerations as well as design.

Additional Information

We have in this chapter attempted to present a useful conceptual principle for measurement of purification. Much productive thinking and experimental investigation which amplify this concept have been accomplished by various researchers. Some articles in the research literature which will be useful for further study of the information presented here are listed in the reference section. Symons, et al. (12) described the use of ΔCOD as a means of measuring the efficiency of treatment, and also outlined some procedures for making treatability studies. The concept of ΔCOD was also described in an article by Hiser and Busch (13). More recently, the concept of ΔCOD as well as overall consideration of biochemical oxygen demand as applied to the treatment plant and the DO profile in receiving streams have been treated extensively by Gaudy (2). Exertion of BOD and methods for the estimation of assimilating capacity of the receiving stream have been discussed by Isaacs and Gaudy (14, 15). Jennelle and Gaudy have studied the effect of dilution of the sample upon exertion of BOD, and have recommended alternative procedures for measurement of BOD (1).

Summary and Conclusions

As a parameter for measuring organic loading or the degree of purification in treatability studies for the design of biological treatment facilities, ΔCOD, i.e., COD_i - COD_e, represents the most straightforward estimate of the amount of biologically available organic matter. Since the BOD test has been used to estimate what ΔCOD actually measures, use of the standard BOD test as a design and operational parameter does not seem warranted. Furthermore, since L_0 (ultimate BOD) can only approach ΔCOD as an upper limit, the ΔCOD parameter gives a more conservative estimate of the ultimate biochemical oxygen demand of the waste sample. For daily operation, ΔCOD is recommended as the primary operational parameter for assessing plant efficiency. Periodic determinations of ΔCOD of the waste in a laboratory reactor allow detection of changes in the nature of the waste or in operational efficiency of the plant.

III

FUNDAMENTAL STOICHIOMETRY OF BIOLOGICAL TREATMENT

"Equation" for Purification

The stoichiometry of purification is often represented by "pseudo-chemical" equations in which the organic matter in the waste is shown as the prime reactant being converted to another form of organic matter (microorganisms) plus CO_2 and H_2O. Most engineers are familiar with this representation:

$$\begin{matrix}\text{Organic}\\\text{matter}\end{matrix} + O_2 \xrightarrow{\text{Microorganisms}} \text{Microorganisms} + CO_2 + H_2O \qquad (3)$$

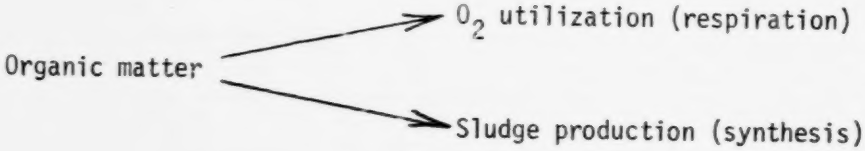
The above equation is a gross simplification of a "chemical" equation (and of microbial growth); it is only qualitative, since no numerical coefficients are included, and significant elements have been omitted. For example, sources of nitrogen and phosphorus and small amounts of many other elements are needed on the left side of the equation, because they are part of the microorganisms produced and are needed both in reactions which form new cellular components and in the reactions by which the waste is degraded. The organic matter in the waste serves primarily as the carbon source, but depending upon the waste in question, it may also contain some nitrogen and phosphorus which are metabolically available to the cells. The microorganisms shown above the arrow are the biocatalysts required to make the reaction proceed.

The reaction is irreversible, although the microorganisms produced are themselves organic matter and could conceivably be used as a source of organic carbon for other microorganisms. Thus, the product can be recycled through the forward reaction but the process cannot be reversed. Reversal of the reaction can be brought about with respect to CO_2 and H_2O as reactants and organic matter (e.g., algae) and O_2 as products through the process of photosynthesis (photo-autotrophic metabolism). This is essentially a reversal of the purpose of biological treatment, and it is now being realized that because of this fact the problems one causes are perhaps equal to or worse than the problems one solves by attempting to rely on photosynthetic oxygen production in biological waste water treatment (e.g., quiescent oxidation ponds). The "swapoff" in organic matter actually conserves organic carbon rather than dissipating it. Conservation of organic matter from waste water sources by such direct recycle may become important as a source of food supply for human beings and lower animals in the long range, but in the intermediate future, at any rate, all modes of disposal of organic carbon in wastes involve processes which in the main lead ultimately to CO_2 and H_2O, which are recycled eventually through the carbon and the hydrologic cycles.

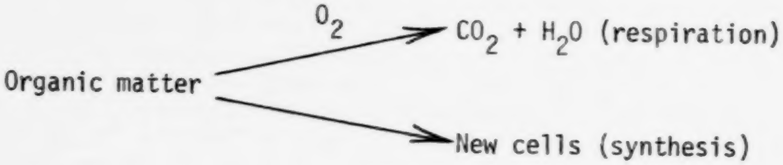
Returning again to the irreversible equation 3, it could be made more mathematically precise if the elements and their combining weights (empirical formulas) for "organic matter" and "microorganisms" (or activated sludge) were known. The chemical composition of the organic matter in the sample is seldom known to such an extent that it can be reduced to a representative chemical formula. There are various empirical formulas which describe the approximate elemental composition of bacterial cells with regard to C, H, O, and N, and these have proven of some use to researchers in analyzing data from investigations wherein readily metabolizable specific compounds of known chemical composition comprised the organic matter being treated. Such a simple stoichiometric chemical equation for purification does represent a simplifying principle; however, it is not easily applied, as a chemical equation, to practical concepts for design of treatment processes. The point to be kept in mind is that the equation represents (as does any other chemical equation) a mass and/or energy balance, and this principle, as we shall see shortly, can be usefully employed to provide quantitative information.

Partition of Substrate Between Respiration and Synthesis

Another representation of purification of wastes is often employed and is perhaps more familiar to water pollution control engineers than is the equation given above. The diagram shown below actually represents the same reactions shown in equation 3, but shows separately the two general types of reactions in which the organic matter participates. That is, it represents the partition of the waste into (1) energy-yielding, O_2-consuming reactions (respiration), and (2) energy-requiring biosynthetic reactions (synthesis of new cells).


Organic matter
O_2 utilization (respiration)
Sludge production (synthesis)

The portion of the organic matter which is used for respiration is oxidized to CO_2 and H_2O; i.e., this portion of the organic matter accounts for essentially all of the O_2 shown on the left-hand side of equation 3 as a reactant and the CO_2 and H_2O shown on the right-hand side as products. Thus, this diagram might be more easily reconciled with equation 3 in the form shown below.



The amount of O_2 which would be required for complete oxidation of the organic material varies with the nature of the compounds present. One quantitative measure of the relative amounts of oxygen required is the respiratory quotient (RQ) which is based upon the relative amounts of CO_2 (totally oxidized carbon) which can be produced from different classes of organic compounds by use of the same amount of O_2. The respiratory quotient, moles CO_2 produced per mole O_2 used, is 1.0 for carbohydrates, 0.8 for proteins, and 0.7 for fats. Thus, for a waste containing primarily carbohydrates, which may be represented as CH_2O, the balanced equation for oxidation (RQ = 1.0) can be written as

$$CH_2O + O_2 \longrightarrow CO_2 + H_2O \text{ (+ xATP)} \qquad (4)$$

For proteins or fats, a proportionally greater amount of O_2 would be required for that fraction which was oxidized.

Another term, ATP (adenosine triphosphate) has been introduced on the right-hand side of equation 4 to represent the amount of chemical energy which was trapped from that released during the oxidation. There are other compounds which organisms can use to trap chemical energy, but ATP is the major one, and the others can be expressed in equivalents of ATP, in any event. It is this trapped chemical energy which now enables the microorganisms to use some of the organic matter (CH_2O, as one example) to make the cellular components needed for new cells:

$$CH_2O + ATP \longrightarrow \text{New microorganisms} \qquad (5)$$

Thus, the diagram and the equation may be combined:

$$\boxed{\begin{array}{l}\text{Organic}\\ \text{matter}\end{array}}\begin{array}{l} + O_2 \longrightarrow CO_2 + H_2O + ATP \text{ (respiration)} \\ + ATP \longrightarrow \text{New cells (synthesis)}\end{array} \qquad (6)$$

The number of moles of O_2 used is approximately equal to the number of moles of CO_2 produced (for carbohydrates) and the O_2 which is used is involved in the complex process of trapping some of the energy released during the oxidation.

Energy Coupling Through ATP

We cannot, in this brief presentation, attempt to explain in simplified terms the mechanics of the processes of biological energy generation and utilization, but a few facts concerning the role of ATP should be mentioned briefly. The precursor of ATP is ADP (adenosine diphosphate). The ATP is thus formed by adding inorganic phosphate, iP, to ADP.

$$\text{ADP} + \text{iP} \longrightarrow \text{ATP} \qquad (7)$$

When the free energy which is contained in ATP is used to drive (to "power") a biochemical reaction requiring energy (a synthesis), the reaction is reversed and ADP and iP are regenerated. It is because of this regeneration of ADP and iP that the phosphorus requirement for cell growth is very low compared to the concentration of carbon source in the waste, and that phosphorus is said to be reused in the system. The ATP thus acts as a "shuttle" which couples oxidation of the waste to synthesis of biological solids.

The efficiency with which microbial cells can use ATP to power synthetic reactions is only approximately known, and it can be expected to vary from species to species. Therefore, a numerical coefficient could not be given for the ATP produced in equation 4. It should be remembered that the reaction by which ATP is made from ADP and iP is a synthesis that requires energy (which is provided from that energy released by the oxidation of the organic matter in the waste). An oxidation of an organic compound in the waste is represented by a loss of an electron. This electron thus "donated" by the carbon source is "accepted" by another compound which in turn passes it along to another until in the final transfer, oxygen accepts the electron and water is formed. A certain amount of dissolved oxygen thus disappears from solution, or one can say that a certain amount of oxygen uptake has occurred. In the metabolic process of passing each electron along from each temporary donor-acceptor system, energy is released in a stepwise fashion and at various points along the way ATP is produced from ADP and iP. The ATP harvest from each passage of an electron from carbon source to oxygen is not the same for all species of microorganisms, nor can it be said with certainty that each time O_2 is used ATP is generated; i.e., oxidation is not necessarily always coupled to ATP generation. Thus the efficiency of the process is not 100%, and efficiency of ATP generation can be expected to vary somewhat from species to species. For this reason, exact numerical coefficients cannot be assigned to any of the terms in the equations given above. This is true even when known pure compounds are metabolized by a heterogeneous population of microorganisms. For wastes of unknown chemical composition, it is even more futile to attempt to theorize regarding quantitative relationships to fit the oversimplified equations used to represent the stoichiometry of treatment. The practical approach to use of these relationships lies in treatability studies, in small-scale pilot plants, in which the quantitative estimates may be based upon actual measurements. Equation 3 then becomes useful as a check on the methods used for measurement.

Energy Balance

Although equation 3 is not suitable for direct use with the majority of wastes (because the chemical formulas are unknown), the sum of the amounts of carbon in the CO_2 and cells produced should balance the amount of organic carbon which has disappeared from the waste due to the growth process (except for wastes containing compounds sufficiently volatile to be stripped in the aerator). Thus, we can make use of the mass and/or energy balance which the equation (or any other chemical equation) represents. One approach requires the capture of all of the CO_2 produced and use of a carbon analyzer to determine the organic carbon content of the waste and of the cells produced. A carbon balance can then be made in which the sum of the carbon produced should be equal to that which has been used.

However, a more readily facilitated method for determining the stoichiometric balance for any specific waste is the energy balance, in which all components of the balance are expressed in terms of oxygen, i.e., as oxygen uptake and COD. If a certain amount of oxygen would be required to completely oxidize soluble organic matter in the waste, and only part of it is oxidized, then the remaining chemical oxygen demand should be equal to the original minus that which has been expressed as O_2 uptake, or:

$$COD_w + COD_{is} = O_2 \text{ uptake} + COD_e + COD_{fs} \tag{8}$$

where COD_w, COD_{is}, COD_e, and COD_{fs} are the COD's of the waste, the initial biological solids, the effluent, and the final biological solids, respectively. Since $COD_w - COD_e = \Delta COD$ as defined in Chapter II, and $COD_{fs} - COD_{is}$ represents the increase in biological solids expressed as COD, or $\Delta COD_{\text{biol. solids}}$, equation 8 may be written as:

$$\Delta COD = O_2 \text{ uptake} + \Delta COD_{\text{biol. solids}} \tag{9}$$

This balance represents the partition of the substrate between respiration and synthesis, in accordance with the diagram shown previously and in accordance with Figure 1, Chapter II.

By returning to Figure 1, we may see the stoichiometric balance depicted graphically. At the end of the purification phase, the amount of organic matter removed is represented as ΔCOD in accordance with concepts presented in Chapter II. The amount of COD which has been oxidized is represented by the accumulated O_2 uptake at this point. This is the amount of "BOD," y, which has been expressed. All of the so-called "BOD" has now been removed from solution. The metabolizable organic matter which has not been oxidized (i.e., expressed as O_2 uptake or BOD) has been incorporated into new biological solids.

By using ΔCOD as the measure of biologically available organic matter in the waste, we have expressed this quantity in useful terms; i.e., in equivalent units of oxygen. Also, O_2 uptake is measured directly in such terms. The increase in biological solids (Δ cells) can also be expressed in terms of oxygen equivalents, either by a calculation or by direct measurement. Using an empirical formula for the cells, the O_2 required to oxidize them completely to CO_2 and H_2O could be calculated, thus yielding a "theoretical COD" of the cells. One such empirical formula is employed below in a balanced equation for total oxidation of the cells to CO_2 and H_2O.

$$\underset{(113)}{C_5H_7NO_2} + \underset{(160)}{5\,O_2} \longrightarrow 5CO_2 + 2H_2O + NH_3 \qquad (10)$$

The ratio of combining weights for cells and O_2 is 160/113 = 1.42; thus each mg of biological solids is equivalent to 1.42 mg O_2, or the calculated COD of the cells = 1.42 x the dry weight of the cells. Alternatively, one can measure the COD of the cells directly by employing the COD test (16).

The mass balance inherent in equation 6, in the diagrams showing partition of substrate, and in the graph of Figure 1, can now be very usefully employed in treatability studies to determine whether the measurements are adequate, i.e., whether all of the organic matter removed from the waste can be accounted for. From equation 9

$$\%\ \text{Recovery} = 100 \times \frac{\text{Accum. } O_2 \text{ uptake} + \Delta\text{COD}_{\text{biol. solids}}}{\Delta\text{COD}} \qquad (11)$$

All measurements may be expressed as mg/l.

There are also other ways in which one may approach the calculation of the materials balance. Various methods, including that described above, have been discussed by Gaudy, et al. (16). However, the energy balance method, equation 11, employing the experimental determination of ΔCOD of the biological solids is straightforward and readily facilitated and is applicable to whole organic wastes of complex and largely unknown chemical composition. Thus there is a relatively easy way to check on the validity of the substrate partition between that which has been dissipated as inorganic carbon (CO_2) in the reactor and that which has been converted to microorganisms (biological sludge). Since these organisms are to be separated in the clarifier, the "BOD" has at this point on the curve of Figure 1 been "purified."

The balance is not absolutely complete, since some of the substrate carbon may be physically stripped from the aerator, depending upon the nature of the original compounds in the waste or the compounds the microorganisms may make during metabolism (17,18). Also, a small amount of the CO_2 produced may be reincorporated into the microorganisms since there are some biochemical syntheses which require a "one-carbon fragment" derived from CO_2, but these possible causes for disturbances in the simple proportion described by equation 9 are not usually of sufficient magnitude to negate the fundamental validity of the principle set forth in the stoichiometric equation or the diagram. The materials and energy balances they imply and the method of making such a balance herein recommended provide an engineer with a relatively easy "bookkeeping" procedure to check the validity of experimental data obtained on the waste under study.

If one is willing to assume that none of the carbon in the waste is unaccounted for, the energy balance given above, equation 9, can be used to estimate oxygen requirements for metabolism. By measuring ΔCOD of the waste and of the sludge, O_2 uptake during the purification phase can be approximated as ΔCOD minus ΔCOD of the biological solids.

The sludge yield, Y, is an important parameter which is usually calculated at the termination of the purification phase. It is best defined as the ratio of the amount of biological solids produced (Δbiol. solids = final - initial) to ΔCOD:

$$Y = \frac{\Delta \text{biol. solids}}{\Delta \text{COD}} = \frac{\Delta X}{\Delta S} \qquad (12)$$

The importance of this parameter is obvious in regard to considerations for subsequent handling of the sludge. The sludge or cell yield will be considered in greater detail in subsequent chapters.

The materials or energy balance could be applied at any point along the curve, either before or after the peak in biological solids and leveling off of the substrate removal curve. The ratio between that which has been respired and that which has been channeled into synthesis (i.e., $y/\Delta X$) at any time may change; for example, it can be seen that to the right of the purification phase in Figure 1 the ratio $y/\Delta X$ increases. However, the balance would still be expected to apply; i.e., ΔCOD is still accountable for largely as O_2 uptake plus the oxygen equivalent of the net amount of biological solids synthesis.

Occurrences which follow the purification phase can be identified as occurrences in the autodigestive or endogenous phase, or the phase of aerobic digestion, and a term accounting for this phase is sometimes included in stoichiometric equations. The term "endogenous respiration" is one which has been used quite often, and it has been subjected to various interpretations. In a strict microbiological sense, it is employed to signify the metabolism of its internal carbon stores by a microbial cell. This type of metabolism may proceed concurrently with metabolism of the external (exogenous) carbon source or after the exogenous substrate is exhausted, or both, but if measured, rather than theoretical, values are used in stoichiometric calculations, this metabolism is included and need not be considered separately.

The most practical way to define "endogenous metabolism" insofar as biological treatment is concerned, is as that metabolism which occurs after removal of the primary substrate, i.e., after the purification phase has ended. It represents respiration (O_2 uptake) at the expense of the accumulated biological solids (see Figure 1); i.e., a net decrease in biological solids concentration. This decrease may be due to oxidation of internal carbon resource in individual cells but, more importantly, with natural populations in waste treatment processes it represents also cannibalization of the microorganisms previously synthesized by predator cells (other bacteria or higher microorganisms such as protozoa, etc.). To the organisms performing the cannibalization, the cells on which they are feeding represent an external, not an internal, carbon source. The stoichiometry of this phase can be written as a grossly simplified chemical equation in which the organic carbon of the cells is oxidized to CO_2 and water:

$$\begin{matrix}\text{Biol.}\\ \text{solids}\end{matrix} + O_2 \xrightarrow[\text{cells}]{\text{Cannibalizing}} CO_2 + H_2O + \text{microorganisms} \qquad (13)$$

Since the cells contain considerable protein which is deaminated during the autodigestive process, some inorganic nitrogen (NH_3) will also appear on the right side of the equation as well as some inorganic phosphorus.

In this phase, cells of one type are converted to cells of another type and to CO_2 and H_2O. The similarity of this equation and equation 3 is obvious, and as stated previously this amounts simply to recycling of the products through the forward reaction with the result that the net solids production is decreased and O_2 utilization and CO_2 production are increased. In short, as the endogenous phase is prolonged, the partition of the initial substrate between utilization for energy production and for synthesis is shifted further toward total utilization for energy. Whether this reaction can approach completion depends upon the ecology of the microbial system, i.e., the right feeding organisms must be present for sequential cannibalization to occur (proper food chain). This important aspect will be discussed in a later chapter on the extended aeration process. It is important here to note that the stoichiometry of this phase is more variable than that of the purification phase; it is less well defined and is, in any event for all but the extended aeration process, a minor consideration in activated sludge processes.

It is important next to consider description of the course of growth during the purification phase to develop useful relationships between growth and substrate removal, since these are phenomena which proceed in the activated sludge reactor and accomplish the real purpose of the activated sludge process.

Summary and Conclusions

Purification proceeds by utilization of the organic matter in the waste for two general types of reactions: (1) oxidation, or respiration, which degrades the organic material to CO_2 and H_2O by oxidative reactions which produce energy in the form of ATP, (2) synthesis of new cells from the organic material by reactions which require energy supplied by ATP. Determination of the fractions of the organic matter utilized for these two purposes is important in treatability studies carried out prior to designing treatment facilities. An energy balance based upon expression of organic matter in the waste and the synthesized sludge in equivalents of oxygen and requiring only measurements of COD and of O_2 uptake is the most readily facilitated method for all types of organic wastes.

IV

KINETICS OF MICROBIAL GROWTH

In the previous chapter, the growth of microorganisms, or synthesis of new cells, was treated from a thermodynamic point of view. That is, the primary concern was determination of the distribution of substrate between energy-producing and energy-consuming reactions, since this distribution corresponds to the portion of organic matter which is effectively removed from the waste (as CO_2 and H_2O) and that which remains as synthesized sludge requiring further disposal. Because sludge production is a primary concern in both design and operation of biological treatment systems, it is important to examine in some detail the kinetics of microbial growth and the factors which control growth and the concomitant removal of organic substrates from the waste.

There are two primary relationships between growth and substrate removal. One relationship concerns the amount of growth (i.e., biological solids accumulation) which one could expect from the utilization of a given amount of substrate. This relationship was defined as the cell or sludge yield in the previous chapter. The second relationship is one between the rate of growth and the concentration of substrate. It should be noted that growth rate can be affected by many experimental or operational factors. When the supply of any required microbial nutrient is limited, it will become the critical factor determining rate of growth or total amount of growth, or both. For example, nitrogen and phosphorus are well known essential nutrients, and when either of these is present in insufficient amount it, rather than the organic carbon source, becomes the growth-limiting component of the waste. Nitrogen and phosphorus are usually added to the waste in sufficient amounts so that they are not factors which exert a determinative effect on growth rate. In short, pollution control engineers usually aim to make the carbon source the growth-limiting nutrient. This principle of design is based upon the use of biological treatment primarily for removal of compounds which exert a biochemical oxygen demand. Consideration of other limiting factors in biological treatment may be necessary to avoid addition of other nutrients to the receiving water, and this aspect will be discussed in Chapter VII. For most organic wastes, however, the carbon source is the limiting nutrient and the most important factor in determining the kinetics of microbial growth during the purification phase. The following discussion will therefore be directed toward the relationship between growth and removal of organic substrates. First, it is important to define exponential growth.

Exponential Growth

In Figure 1 it may be observed that the course of substrate removal takes the approximate form of an inverted S-shaped curve and that the curves for O_2 uptake and biological solids concentration (which depict

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the "fate" of the removed substrate) also follow S-shaped curves. The S-curves are not necessarily symmetrical, and in most cases they are not. For example, the inflection point in the biological solids (X) curve usually will be located (as shown in Figure 1) somewhat higher than the mid-point of the curve. The inflection point in the O_2 uptake curve may or may not come at about the same relative position as that for X. Generally, an inflection point can be observed in the substrate curve in a position approximately directly above the one in X. The precise locations of all of these inflection points are not overly important, but it is emphasized that when such an experiment as shown in Figure 1 is run in the laboratory or in nature, both the disappearance of the initial growth resource (in this case, the carbon source) and the increase in biological solids usually proceed at an increasing rate, followed by a period of decreasing rate, until the resource is depleted. That is, in such an experiment as that shown, one kinetic order does not prevail throughout the course of the experiment whether one examines substrate removal, O_2 uptake, or cell growth.

The period of increasing rate of growth can very often be described mathematically as one in which X increases exponentially. Such a kinetic situation can be expressed in the form of a first order differential equation:

$$\frac{dX}{dt} = \mu X \tag{14}$$

The rate of increase in X, i.e., dX/dt, is proportional to the amount of X present at any instant. The proportionality constant, μ, is known as the specific growth rate, or the exponential growth rate constant.

Equation 14 is a useful description of growth, but is more valuable in its integrated forms:

$$X_t = X_o e^{\mu t} \tag{15}$$

or

$$\ln X_t - \ln X_o = \mu t \tag{16}$$

From the latter form it can be seen that a plot of ln X vs. t would form a straight line of slope μ.

In Figure 2, the biological solids (X) curve of Figure 1, up to the point of maximum solids accumulation, is replotted on both arithmetic and semi-logarithmic coordinates. It can be seen that the extent

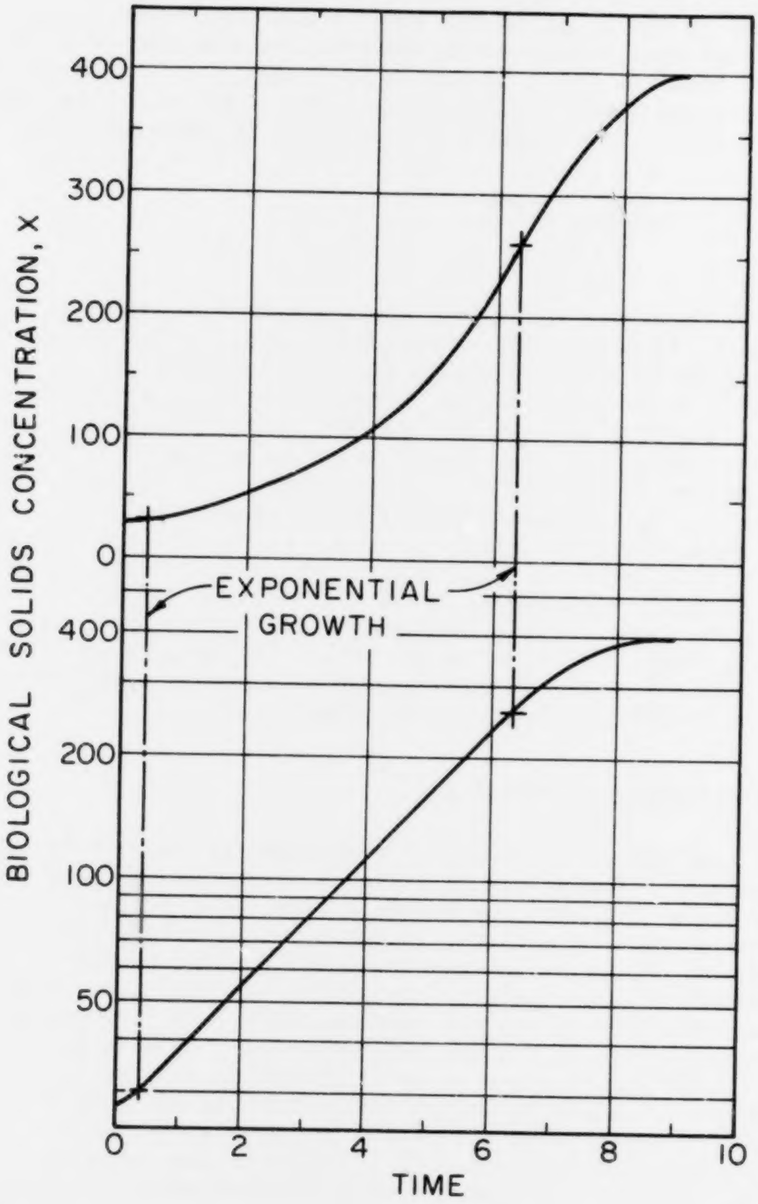

Figure 2. Arithmetic and semi-logarithmic plots of microbial growth during the substrate removal phase in a batch system.

(duration) of the exponential phase of growth can be assessed by noting the extent of the straight line portion of the semi-log plot. The end of the exponential phase corresponds to the point of inflection on the arithmetic plot. The numerical value of μ can be determined as the slope of the line in accordance with equation 16, but is most easily computed by determining from the semi-log plot the time required for X to double in value (i.e., t_d) by extrapolation, if necessary, since the exponential phase may be shorter than one doubling time. Substituting these values in equation 16, μ is defined as:

$$\mu = \frac{\ln 2}{t_d} = \frac{0.693}{t_d} \tag{17}$$

It is important to examine the concept of exponential (logarithmic) growth, since various interpretations and significances have been assigned to "log" growth by basic microbiologists and by engineers as well. It is important to remember that, regardless of any particular physiological significance of this phase of growth, it is unequivocally, by definition, growth which is mathematically describable by equations 14 through 17. It is that growth for which the doubling time t_d, is constant regardless of the magnitude of t_d or of μ. It is that period of growth for which one's data will be described by a straight line on a semi-logarithmic plot of X vs. t. In short, logarithmic or exponential growth is simply a mathematical description, and the symbol μ is a proportionality factor which is empirically determined from experimental data. The importance of the mathematical definition of exponential growth will become apparent when continuous flow cultures are considered (Chapter V).

Relationship Between μ and S

We have defined μ simply as a mathematical descriptor which is a rate constant allowing one to assign a numerical value to the rate of increase in X when the doubling time for X is constant. There are some workers in both the fields of microbiology and biological engineering who maintain that the doubling time cannot be constant, and therefore logarithmic or exponential growth cannot exist, unless the substrate, and all other required nutrients, are present in excess concentration (e.g., see reference 19). This is an old concept which has been passed along as an axiom for so many years that it has become dogma. We state here that the concept is false, and we will not burden the reader with all of the proven reasons for our statement. A more detailed discussion is provided in the research literature (20, 21). We will state here simply that the weight of experimental data indicates that exponential growth identifiable by specific values for μ, its descriptor, can occur at concentrations of substrate substantially below those for which μ approaches some maximum value at which addition of higher concentrations of substrate causes no further increase in μ.

This statement can be verified by a relatively simple experiment. If one were to run a series of growth experiments concurrently in separate reaction flasks into which he placed small inocula of cells acclimated to a particular substrate, and in each reaction flask he had placed increasing concentrations of the substrate covering a wide range of concentrations, he would observe growth curves (arithmetic plots of X vs. t) which would be different in two respects. First, he would find that the more substrate he had in the flask, the more total growth (i.e., higher concentration of X) he would have at the end of the substrate removal phase, i.e., the increase in biological solids should vary with initial substrate and the proportionality factor, Y, should be constant. He would also observe that the rate at which X attained its final value would be lower for the systems in which he had placed the lower substrate concentrations. He would also observe that the growth curves would become more alike and would become practically superimposable for the flasks into which he had placed higher and higher substrate concentrations.

If one now plotted the growth curves on semi-logarithmic coordinates (i.e., ln X vs. t), the data for the period up to the inflection point on the arithmetic curves would plot as straight lines, the slope increasing with increasing initial substrate concentration, until a maximum slope was attained. Such an hypothetical experiment is shown in Figure 3. From this figure, one observes an apparent dependence of the value of μ on the initial substrate concentration in which the μ was generated. The mathematical description of this apparent relationship may be examined initially by plotting the values of μ against the corresponding values of initial substrate concentration, S_o. Such a plot is shown in the upper portion of Figure 4. It is seen that as the initial substrate concentration approaches higher and higher values, μ approaches a maximum value (μ_{max}) and further increases in initial substrate concentration cause no further increase in μ; i.e., the value for μ eventually becomes independent of substrate concentration.

It has been observed that most growth data of this type can be satisfactorily fitted with a hyperbolic function, as given below:

$$\mu = \mu_{max} \frac{S_o}{K_s + S_o} \tag{18}$$

This equation (the "Monod equation") has appeared in innumerable places in both the microbiological and the pollution control literature with the symbol S instead of S_o, and such "familiarity" is in large measure responsible for the varying "theoretical" interpretations which have been placed upon equation 18. These aspects are described elsewhere (20, 21). It is important here to note that the equation is a useful relationship which was derived empirically and is simply an analytical

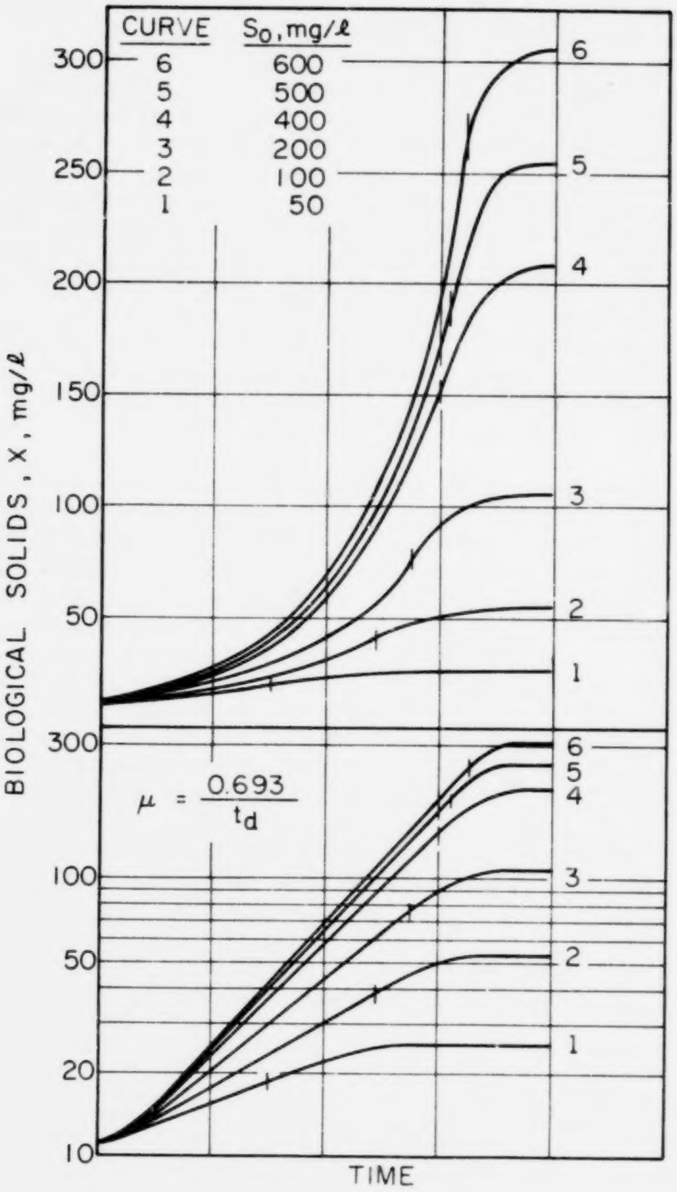

Figure 3. Effect of initial substrate concentration, S_0, on the rate and total amount of microbial growth. The vertical lines indicate the end of the exponential growth phase.

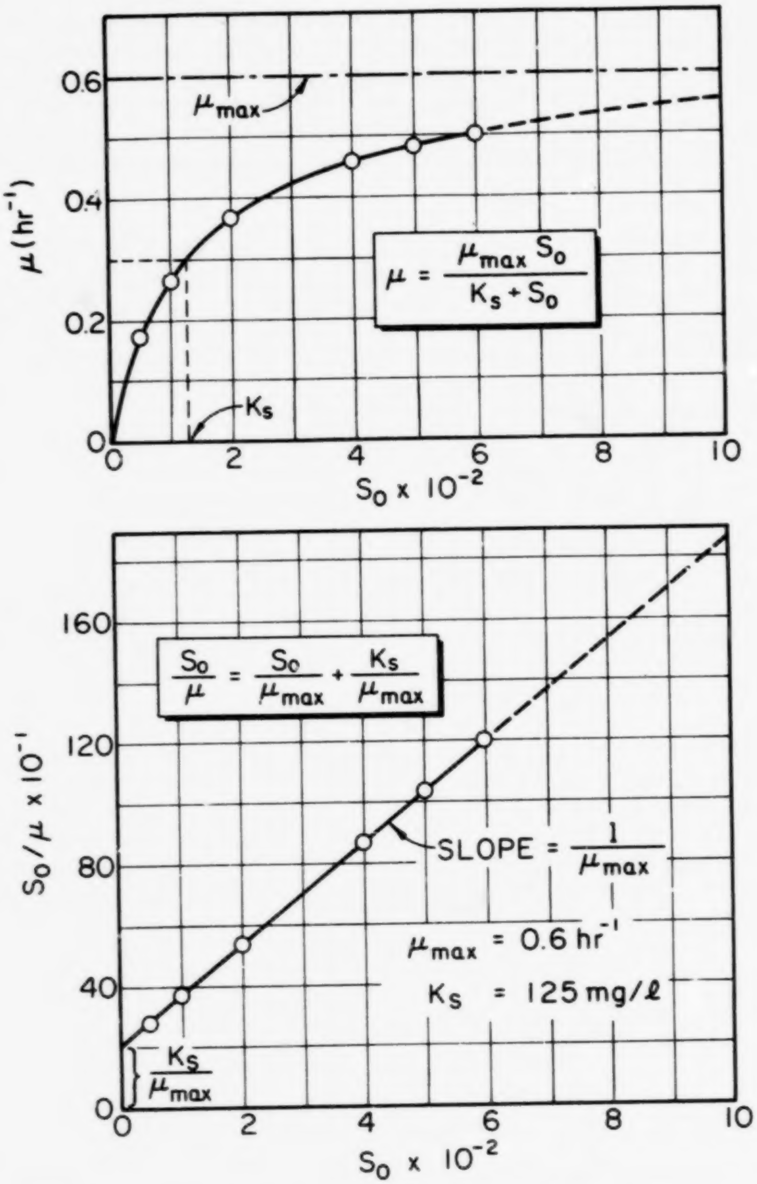

Figure 4. Hyperbolic plot of the relationship between specific growth rate, μ, and initial substrate concentration, S_o (upper graph), and a straight line plot, S_o/μ vs. S_o, of the same data (lower graph).

expression which has been found to fit a large amount of experimental data. Like μ_{max}, the term K_s is a constant in the equation. Whereas μ_{max} represents the constant value of μ which is not exceeded in the presence of higher substrate concentrations, K_s (the "saturation" constant) determines the sharpness of curvature of a plot of μ vs S_o. The higher the value of K_s, the flatter will be the curve and the more slowly will μ become asymptotic to μ_{max} as substrate concentration is increased.

This behavior of the expression is shown in Figure 5 (taken from reference 22) in which are plotted curves for μ calculated from equation 18 for a range of values of S_o. For each curve, the indicated value was used for K_s and μ_{max} for all curves was 0.4 hr^{-1}. It can be seen from the figure and equation 18 that if K_s were infinitely small, there would be very little dependence of μ on S, and for all practical purposes only one exponential growth rate constant (μ_{max}) would describe growth. When K_s is very small, the curve rises very sharply and breaks to the right very sharply. In this instance, μ can be said to be very sensitive to a small change in substrate concentration at very low concentrations only. For systems which exhibit a very low value of K_s, the sharply rising portion of the curve could possibly be fitted to a straight line which intersected the horizontal asymptote, μ_{max}. However, there are ample data in the literature (22-26) to indicate that the relationship between specific growth rate and substrate concentration is such that it can be most easily fitted to the continuous hyperbolic function, equation 18.

It can be seen from equation 18 that the substrate concentration at which μ is equal to half the value of μ_{max} is the numerical value of K_s. Thus from a plot of the data (μ vs. S_o) one could estimate μ_{max}, then locate the intersection of 0.5 μ_{max} with the curve and read the K_s value on the abscissa (see Figure 4). A better procedure is to arrange equation 18 in one of its straight line forms. Two of these forms are given below:

$$\frac{1}{\mu} = \frac{1}{\mu_{max}} + \frac{K_s}{\mu_{max} S_o} \qquad (19)$$

$$\frac{S_o}{\mu} = \frac{S_o}{\mu_{max}} + \frac{K_s}{\mu_{max}} \qquad (20)$$

In the lower portion of Figure 4, the μ and S_o values shown in the upper portion are plotted in accordance with equation 20. These forms of equation 18 simply allow one to make a more accurate estimate of the values of the kinetic "constants" μ_{max} and K_s because one usually has only a small number of values of μ in the low substrate ranges, and by using a straight line form one is not required to make a graphical estimate of curvature.

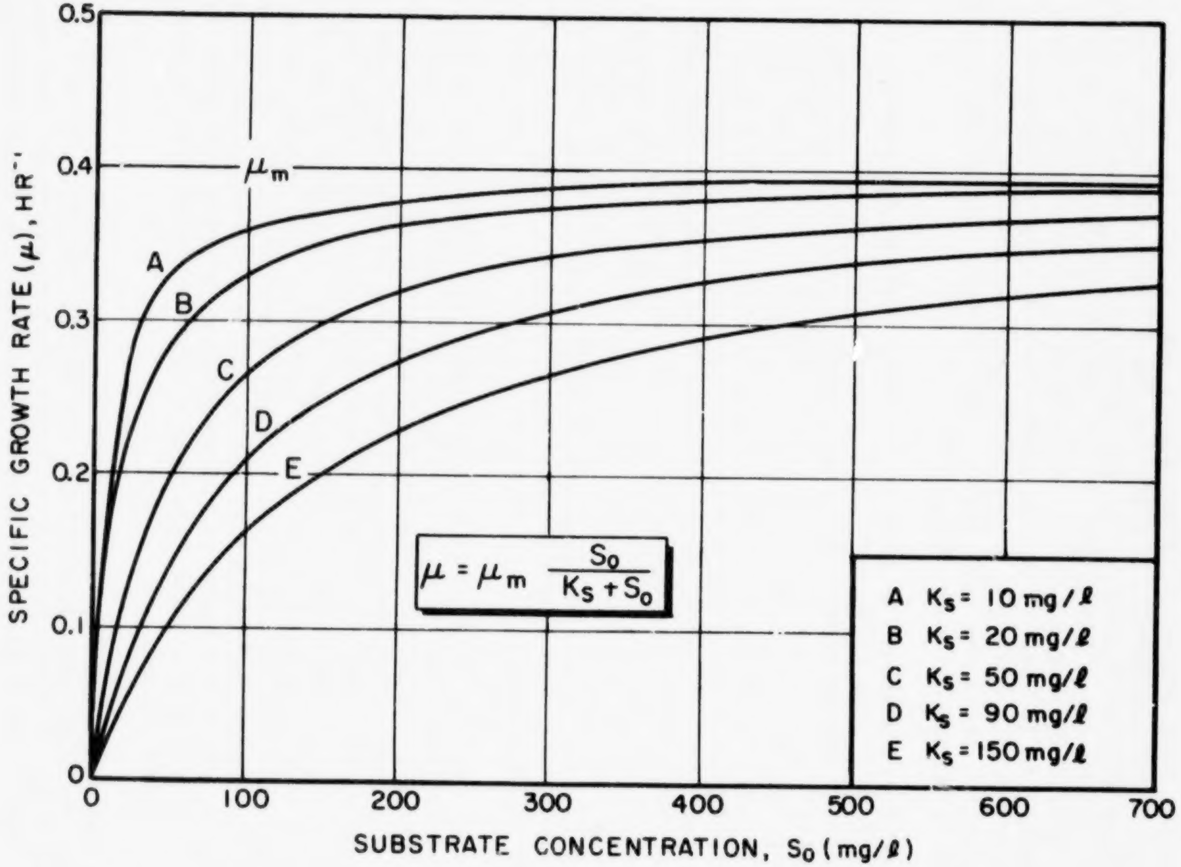

Figure 5. Effect of the numerical value of the saturation constant, K_s, upon the degree of curvature of the hyperbolic relation between μ and S_0. Curves were calculated using equation 18 with $\mu_{max} = 0.4$ hr^{-1} and the values of K_s shown on the figure (22).

Equation 18 has been employed by biochemical (fermentation) engineers and microbiologists, most of whom deal with pure cultures of microorganisms. For any specific pure culture, μ_{max}, K_s, and Y are constants for that organism under constant cultural conditions. In the activated sludge process, natural microbial populations exhibiting a rather high degree of heterogeneity and variability are used (anything but "pure" cultures). Values for the kinetic constants have been shown with pure cultures to vary with species, substrates, temperature, pH, etc.; therefore, in the pollution control field one must evaluate these constants in repetitive experimentation in order to determine if there is a reasonably narrow range of variation for engineering use. Also, there could be some valid doubt as to whether equation 18 is the most logical one to employ for relating μ and S_o for systems such as activated sludge.

For these reasons, we carried out many experiments over a considerable period of time using different heterogeneous populations developed from various sewage seeds. All experiments were run at 25 ± 2°C and at neutral pH. A synthetic waste was used in all experiments, and a number of different carbon sources of different types were used as substrates. However, in order to determine the variability in the constants due only to differences in the populations used in different experiments, a large number of repetitive runs were made with a single substrate. Glucose was used in these experiments, since it is easily metabolized by most bacteria and should allow the greatest heterogeneity possible in a microbial population growing on a single organic compound. Values for μ_{max} and K_s determined in twenty-two separate series of growth experiments using glucose as carbon source are shown in Table I.

As these data show, even under constant conditions and with a simple synthetic waste, there is no single value which can be used as "the" kinetic constant for a heterogeneous population. Values obtained for these constants with other substrates were also variable (26), but only a few experiments were made with each and the range obtained with glucose is probably more representative of the range to be expected with heterogeneous populations at moderate temperatures and neutral pH. Based upon the values given in Table I and upon other experiments, it would appear that ranges of 0.4 to 0.6 hr^{-1} for μ_{max} and 50 to 125 mg/l for K_s might be chosen as representing reasonable estimates for use in applying these kinetic constants to heterogeneous populations such as activated sludge.

In all of these studies in which μ_{max} and K_s were evaluated for different heterogeneous populations, the fit of the data to the hyperbolic equation of Monod (equation 18) was also determined. This equation (using initial substrate concentration, S_o) was found to provide a more satisfactory fit of the experimental data than did other equations and relationships tested with the same data (22, 27).

Municipal sewage represents, perhaps, one of the most complex mixtures

TABLE I

VALUES OF μ_{max} AND K_s FOR HETEROGENEOUS POPULATIONS OF SEWAGE ORIGIN GROWING ON GLUCOSE

μ_{max} hr^{-1}	K_s mg/1	μ_{max} hr^{-1}	K_s mg/1
0.49	29	0.40	93
0.38	11	0.72	105
0.42	68	0.50	108
0.38	87	0.45	18
0.59	91	0.63	13
0.72	145	0.63	58
0.56	97	0.48	52
0.77	181	0.40	58
0.60	116	0.74	111
0.53	30	0.31	23
0.44	37	0.46	85

Averages: $\mu_{max} = 0.53\ hr^{-1}$, $K_s = 74$ mg/1.

of carbon sources found in nature. It is such a weak substrate that it is difficult to use in growth experiments requiring a gradation of initial substrate concentration. Therefore, the soluble portion of a municipal sewage was concentrated in a flash evaporator, and growth on various dilutions of this carbon source was studied (26). Values of μ were determined from semi-logarithmic plots of optical density vs. time, and values for one such experiment are plotted vs. the corresponding substrate concentrations (measured as COD) in Figure 6. Average values for sewage were $\mu_{max} = 0.46\ hr^{-1}$ and $K_s = 52$ mg/l. These are in general accord with the range previously given for glucose. Data such as these provide some assurance that these relationships indeed do exist for complex populations and for complex substrates, and that there is a reasonably narrow range of expected values.

In the pure culture area, very little experimentation regarding the curvature factor (K_s) of equation 18 has been done. It is generally believed that the value of K_s will be much below 50 mg/l of substrate, largely because in the early work by Monod (23) low values were found. It is often overlooked that these studies involved only two bacterial species and that other early studies yielded a very high value for K_s (24). These aspects and their ramifications for use of equation 18 in research need not be dealt with here but are covered elsewhere (20, 21). However, before passing on to another important "kinetic constant" it is appropriate to mention that the analytical geometry of equation 18 is the same as that of the Langmuir adsorption isotherm and that of the Michaelis-Menten equation for dependence of enzyme velocity on substrate concentration. It is necessary to point out that similarity in kinetic form does not imply similar mechanism.

The point to be emphasized here is that equation 18 is usable as a kinetic description for determining the effect of initial substrate concentration on μ. It is an entirely empirical one which can be recommended for use because it provides a good fit to growth data. It will be used in a later chapter in the development of a kinetic model for describing biological behavior in a continuous flow reactor, i.e., an activated sludge tank.

Sludge Yield

In the previous chapter, sludge yield was defined as the amount of biological solids produced per unit of organic material used. Within the context of the mass balance equation, it is therefore most commonly determined at the completion of the reactions, i.e., at the point of maximum biological solids accumulation, which normally corresponds to the point of termination of substrate removal. It is important to note that while its determination at the end of the substrate removal phase is valid, the factor Y is applicable throughout the growth curve, i.e., during the exponential and the declining phases of growth. Thus Y is useful as a kinetic constant, and we can write

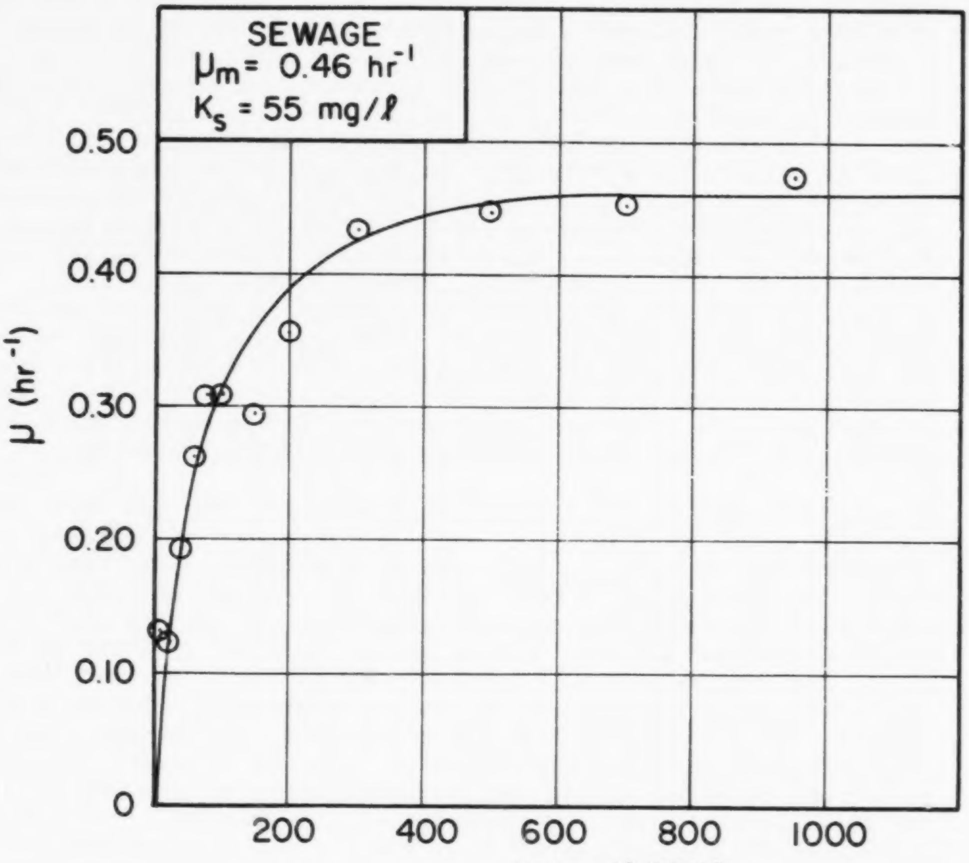

Figure 6. Hyperbolic plot of the relationship between specific growth rate, μ, and initial substrate concentration, S_0, for a heterogeneous microbial population of sewage origin growing on a concentrate prepared from the soluble portion of municipal sewage (26).

$$\frac{dX/dt}{dS/dt} = Y \qquad (21)$$

In large measure, the above statement is true and can be shown to apply to heterogeneous microbial populations. The relative constancy of sludge yield throughout the substrate removal period is demonstrated in Figure 7. These results were obtained in an experiment in which glycerol at relatively high initial concentration was fed to a heterogeneous population at low initial solids concentration (28). Biological solids and filtrate COD concentrations were measured and constitute the primary (raw) data shown in Figure 7A. The substrate removal curve was constructed by subtracting the COD curve from the initial COD. In Figure 7B, this curve and the biological solids curve are plotted on semi-logarithmic coordinates, and in Figure 7C, the substrate removal curve is plotted vs. the biological solids curve. It is apparent that the ratio of solids produced to substrate removed, i.e., $\Delta X/\Delta S$, is relatively constant throughout the substrate removal period. The calculated correlation coefficient, r, is 0.975. The sludge yield determined in the usual way at the end of the substrate removal period is thus also applicable to the exponential phase of growth; i.e., Y is a constant during any particular experiment.

It should be emphasized that this constancy of Y does not imply that one would obtain precisely the same value for the next experiment. Sludge yields, like μ_{max} and K_s, can be expected to vary, depending largely on the species of cells present and upon the substrate. The substrates one encounters in waste waters are, except for certain highly specific industrial wastes, not so selective that one could expect the same species to predominate all of the time; therefore, Y will vary not only from waste to waste (substrate to substrate) but even for the same substrate at different times, and we must seek to determine a range of expectable values. Thus it is important to orient our thinking, not to precise constants based upon theoretical reasoning (energy trapping, thermodynamics, energetics, etc.), but to a probable range of values which will be observed for naturally developed populations. This is, incidentally, the type of thinking which engineers should be readily able to perceive. It may be more convenient to use one value, but it is unrealistic to do so for the natural biological populations with which pollution control engineers must deal. For an engineer to accept a certain value for a biological "constant" to two significant places is as much of a mistake on his part as is the insistence on the part of some basic microbiologists that valid conclusions can be drawn only from studies in which pure cultures were used. The engineering mechanics of the bio-mass, which is indeed a rather new engineering material, are just now beginning to evolve. At this stage of the development of this vital engineering field, one cannot afford to replace empiricism and experimentation with rather weakly based simplifying theory. The importance of microbial ecology and the fact that shifts in predominance continually

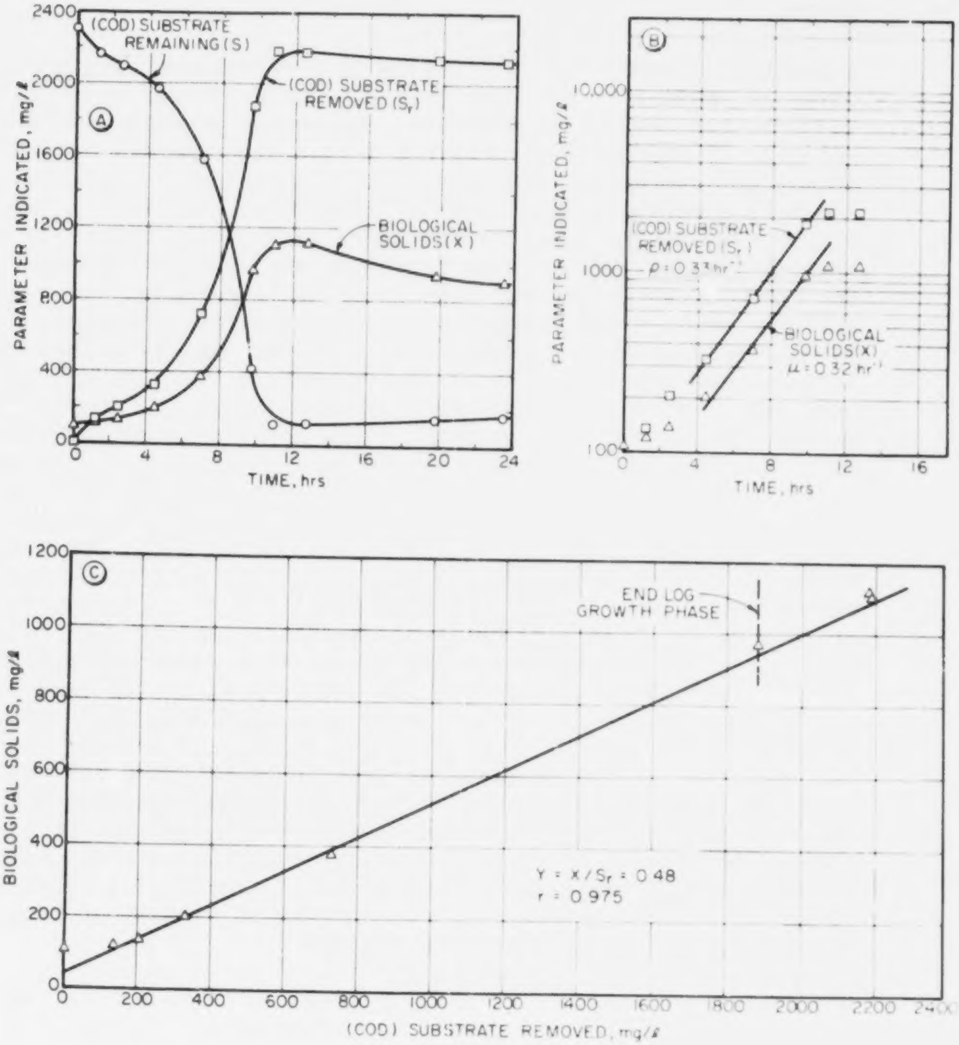

Figure 7. Constancy of sludge yield throughout the substrate removal period during growth on glycerol of a heterogeneous microbial population of municipal sewage origin (28).

occur, mean that there is an inbuilt variability in the properties of the bio-mass, and the engineering profession has to gain some definitive insight regarding the range of variability and its effect on working theories for design and operation before it can embrace any theory.

Statistical ranges for cell yields on a variety of carbon sources are given in Table II. For more in-depth discussion of cell yield in heterogeneous microbial populations, the reader is referred to recent articles in which values obtained in our laboratories during the last decade are summarized (28, 29). It is important to note that most of the sole carbon sources shown in Table II do not exert a very selective effect on the population. Many types of organisms can metabolize these compounds. All of the organisms employed in the many studies for which Y was determined were grown up (acclimated) on the compound tested from original sewage seeds all taken from the same municipal sewage treatment plant (Stillwater, Oklahoma). All values were obtained under the same operational (experimental) conditions. All values were measured in the same manner, at the end of the substrate removal period (see Figure 1). In cases where eight or more experiments were run, the standard deviation, coefficient of variance (CV), and 95% confidence limit (CL) were calculated. For example, the average yield, $\bar{Y}$, in twelve experiments employing lactose was 47.1%. The lowest yield obtained was 30, and the highest, 61%. The standard deviation of the mean was 7.5%, i.e., two-thirds of the Y values were between 39.6 (47.1 - 7.5) and 54.6 (47.1 + 7.5). Expressed as a percentage of the mean Y value (i.e., expressed as the coefficient of variance, CV), two-thirds of the values fell within ± 15.9% of the mean Y of 47.1, and there is a 95% chance that if the experiments were run again, the $\bar{Y}$ would fall between 42.4 and 51.8% (95% CL).

The fact that a wide range of values can be obtained for any given substrate attests to the variability of the bio-mass and to the engineering need to work within a range of reasonably expected values, i.e., a statistical range. Data such as these and other data as well (28, 29) indicate that a reasonable range of $\bar{Y}$ for engineering calculation is 40 to 60% for most carbon sources consisting largely of sugars and sugar alcohols and combinations thereof.

The cell yield can also be obtained from continuous flow reactor data and although we have detected some differences in average values (27, 29) for batch and continuous flow systems, most fall within the range cited above. Also, while there is some evidence that reactor detention time in continuous flow processes may exert some effect on $\bar{Y}$ (i.e., somewhat lower values may be observed at very high detention times), the data available indicate that at reactor detention times below 24 hours, the effect is not readily observable.

TABLE II

STATISTICAL SUMMARY OF SLUDGE YIELD VALUES FOR HETEROGENEOUS POPULATIONS OF SEWAGE ORIGIN GROWN ON VARIOUS CARBON SOURCES

Carbon Source	$\bar{Y}$	n	Range	S	CV	CL
Arabinose	44.8	6	32-51	-	-	-
Cellobiose	50.3	4	34-61	-	-	-
Fructose	53.0	8	34-69	10.4	19.6	44.4-61.6
Galactose	51.9	24	36-76	11.3	21.8	47.1-56.7
Glucose	61.9	118	36-88	12.5	20.1	59.6-64.2
Glycerol	46.5	31	31-61	9.4	20.2	43.1-49.9
Lactose	47.1	12	30-61	7.5	15.9	42.4-51.8
Maltose	51.7	7	39-86	-	-	-
Mannose	52.2	8	36-70	13.4	25.6	41.1-63.3
Ribose	45.7	9	36-56	6.4	14.0	40.8-50.8
Sorbitol	50.6	39	35-59	6.0	11.8	48.7-58.5
Sorbose	52.0	12	23-73	16.7	32.1	41.4-62.6
Sucrose	53.1	12	33-77	14.1	26.5	44.2-62.0
Xylose	50.4	18	25-71	11.4	22.6	44.8-56.0

Y is expressed as percent substrate used for synthesis, i.e., 100 x $\Delta X/\Delta S$.

Summary and Conclusions

In this chapter we have considered the relationship between substrate removal, the desirable result of biological treatment, and microbial growth, which accompanies substrate removal. Each species of microorganism is characterized by an absolute maximum growth rate which can be achieved under optimum conditions with all nutrients in excess. Under these conditions, growth is exponential and is described mathematically as

$$\frac{dX}{dt} = \mu X \tag{14}$$

At maximum growth rate, $\mu = \mu_{max}$.

Growth rate may be limited at values below μ_{max} by sub-optimal conditions of temperature, pH, aeration, etc., or by limiting concentration of any required nutrient. Since the objective of biological treatment is removal of the organic carbon source with the greatest possible efficiency, the process is normally designed so that the carbon source is the limiting nutrient, e.g., by addition of nitrogen and phosphorus, if necessary. Therefore, the specific growth rate, μ, becomes a function of substrate concentration, i.e., $\mu = f(S)$. With substrate concentrations below those which allow growth at the maximum rate for the cultural conditions used (pH, temperature, etc.), exponential growth occurs, i.e., growth which plots as a straight line on semi-logarithmic coordinates. The proportionality constants (μ) for exponential growth can be evaluated from the slopes of such plots of X vs. t for various initial substrate concentrations using equation 17

$$\mu = \frac{0.693}{t_d} \tag{17}$$

where t_d is the time required for a doubling in mass. A plot of these values of μ vs. initial substrate concentration for heterogeneous microbial populations has been shown to fit a hyperbolic function, the "Monod equation" (equation 18).

$$\mu = \frac{\mu_{max}\ S_o}{K_s + S_o} \tag{18}$$

Evaluation of the kinetic constants, μ_{max} and K_s, may be made using one of the straight-line forms of the hyperbolic equation (e.g., equation 19 or 20).

For heterogeneous populations, affected by ecological shifts which cause changes in predominating species, precise values for the kinetic "constants" cannot be expected. However, repeated determinations of these constants for populations selected from municipal sewage have demonstrated that a usable range of values can be defined for such populations. For heterogeneous populations of sewage origin, a reasonable range for μ_{max} (at neutral pH and 25 ± 2^oC) appears to lie between 0.4 and 0.6 hr^{-1} and K_s values are generally between 50 and 125 mg/l. The fact that values within the same range were found for growth on the soluble portion of municipal sewage lends added credence to these results. Values outside these ranges may be observed, but the ranges appear adequate to cover most situations in which a readily metabolized carbon source is the limiting nutrient for natural populations at approximately neutral pH and moderate temperatures.

The rate of growth, dX/dt, is also related to substrate concentration by the following expression:

$$\frac{dX}{dt} = \frac{dS}{dt} (Y) \qquad (21)$$

The kinetic "constant," sludge (cell) yield, Y, applies in general throughout the growth cycle and substrate removal period, i.e., the exponential, or logarithmic, phase and the declining phase of growth. Its determination is usually most easily facilitated by determining the amount of biological solids produced and substrate (COD) removed at the end of the substrate removal period:

$$Y = \frac{\Delta X}{\Delta S} \qquad (12)$$

Generally, expectable values for Y will lie between 0.40 and 0.60 (40-60%).

Thus, during exponential growth, both the rate of accumulation of solids and the total amount of biological solids synthesized are relatable to the concentration of carbon source in definable fashion, which can be quantitatively estimated by three biological systems constants, μ_{max}, K_s, and Y.

In the following chapter, these biological kinetic constants which are determinable in batch systems, are to be related to physical and hydraulic characteristics of continuous flow systems, e.g., activated sludge.

V

KINETICS OF COMPLETELY MIXED REACTORS

It is a well known fact that throughout much of its history, the activated sludge tank has been designed as a long, narrow tubular reactor through which the waste passes under aeration. The reactor has usually been considered as a plug, or piston, flow system into which a waste containing a relatively high concentration of organic carbon enters, and from which a low concentration of organic carbon exits at the effluent end. Thus X and S are envisioned as changing, X increasing and S decreasing, from influent to effluent ends.

It has long been known also that this does not actually happen, because such perfect piston flow is extremely difficult to obtain. The longer and narrower the tank, and the less turbulent the mixing, the closer will such a hydraulic regime be approached. It is roughly approximated in some rivers. It is difficult to attain in laboratory growth reactors, and seldom, if ever, does it obtain in activated sludge tanks.

At the other end of the spectrum of hydraulic regimes which are theoretically possible in biological reactors is one in which the inflowing material is instantaneously and homogeneously, i.e., completely, mixed with the liquid and suspended particles in the reactor. While perfect plug flow is very difficult to attain, conditions approaching perfect complete mixing are less difficult to achieve. Since even activated sludge tanks which are not designed as such are in the main closer to complete mixing than to plug flow, the kinetic aspects of the former are of most critical importance in considering the activated sludge process.

We can say with considerable certainty that growth of heterogeneous microbial populations in batch systems does, in general, follow the "rules" covered in the previous chapters. In this chapter we will examine the behavior of microbial populations in a completely mixed reactor under continuous flow conditions.

Criteria for Complete Mixing

A simple reactor of the continuous flow, once-through type is diagrammed in Figure 8. The batch reactor has been converted to continuous flow by the continuous pumping of medium into the tank at a constant rate of flow, F, and concurrent withdrawal by overflow at the same rate so that the volume of mixed liquor, V, is constant. The inflowing concentration of carbon source is S_i. The concentrations of substrate and cells in the reactor and in the reactor effluent are S and X, respectively. If the reactor is completely mixed, the concentrations of substrate and cells are the same in all parts of the reactor; therefore, these concentrations in the mixed liquor exiting the reactor

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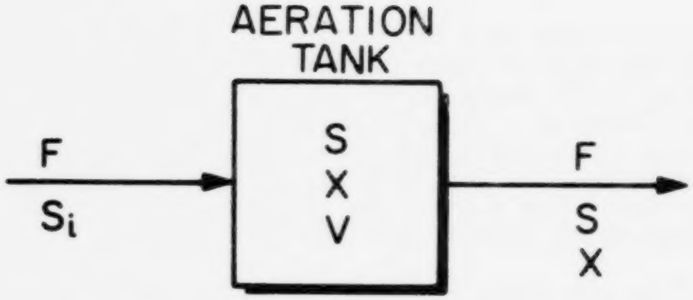

$$D = F/V = 1/\bar{t}$$

Figure 8. Flow diagram for a continuous flow, completely mixed reactor of the once-through type.

are also the same. Also, if the reactor is completely mixed, the incoming substrate is "instantaneously" mixed; i.e., diluted in the reactor, the dilution factor being the ratio of the volumetric rate of inflow, F, to the volume of reaction liquor in the aeration tank, V. Thus the dilution factor, or the dilution rate, D, is equal to F/V, and D is the reciprocal of $\bar{t}$, the mean residence time (detention time) in the reactor. This hydraulic parameter is an extremely important one, as will be shown below.

Before the kinetics of completely mixed systems are discussed, it is necessary to consider ways to ascertain whether a reactor is, or is close to being, completely mixed. The criterion of the sameness of X (biological solids concentration) in the reactor and the reactor effluent can be checked (approximated) by measuring the transmission of light (OD, optical density) by the mixed liquor in the reactor and in the reactor effluent. The optical density provides a quick estimate of biological solids concentration. In cases where X is so high that correlation between optical density and biological solids is not possible, the samples of reactor mixed liquor and effluent can be equally diluted before the comparison is made. Also, for best results the floc must be vigorously dispersed before reading optical density. The criterion of complete mixing with respect to substrate could be checked by sampling at the same places, but this is more time-consuming and, in any event, if the mixing in the reactor is vigorous enough to satisfy the criterion with respect to suspended particles (X), it can be assumed that the criterion will also be satisfied with respect to substrate molecules (S).

Complete and instantaneous mixing characteristics of the reactor can also be checked by performing either dilute-in or dilute-out (washout) studies. A washout study could be performed by filling the reactor tank to the operating volume with water containing a known concentration, C_0, of dye (for example), or suspended particles (e.g., clay). If clear water is now pumped into the tank and the concentration in the effluent is measured, it can be seen that, if complete mixing obtains, the concentration will be continually decreased (diluted) by the factor F/V, i.e., D. Thus we may write

$$\frac{dC}{dt} = -DC \tag{22}$$

or, after integration:

$$C_t = C_o e^{-Dt} \tag{23}$$

If, on the other hand, the reactor had been filled with clear water and at zero time water containing dye, clay, or another marker material at C_0 had been pumped into the tank, the concentration C_t in the effluent at any time thereafter would be given by the dilute-in equation:

$$C_t = C_o(1 - e^{-Dt}) \tag{24}$$

Theoretical dilute-out and dilute-in curves calculated for a dilution rate of 0.125 hr^{-1} are shown in Figure 9.

A comparison of theoretical and observed dilute-in curves obtained for one of the laboratory-scale continuous flow reactors employed in research in our laboratories is shown in Figure 10. Experimental values for two dilution rates, 1/4 and 1/8 hr^{-1}, are shown. Circular, well-agitated tanks of considerable depth in relation to surface area appear to possess the geometry favorable to complete mixing.

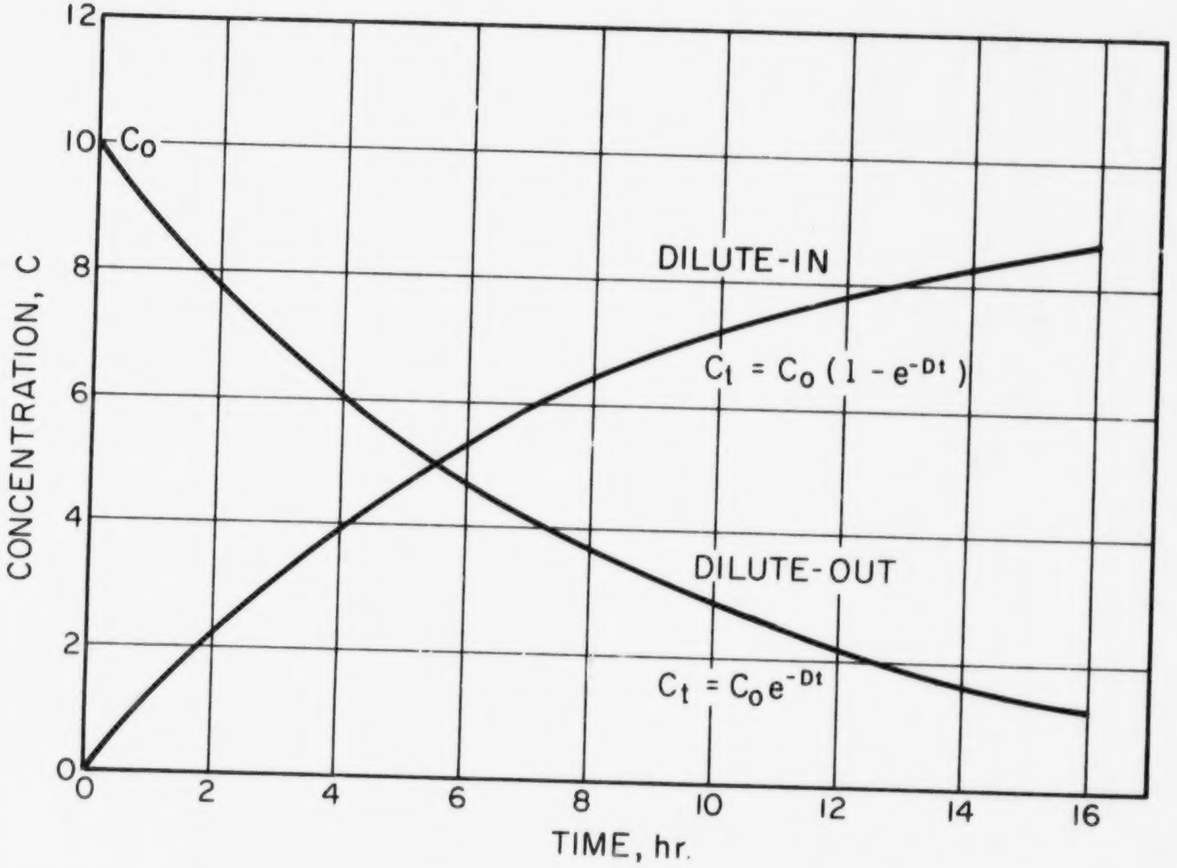

Figure 9. Theoretical dilute-in and dilute-out curves for a completely mixed reactor calculated for a dilution rate of 0.125 hr^{-1}.

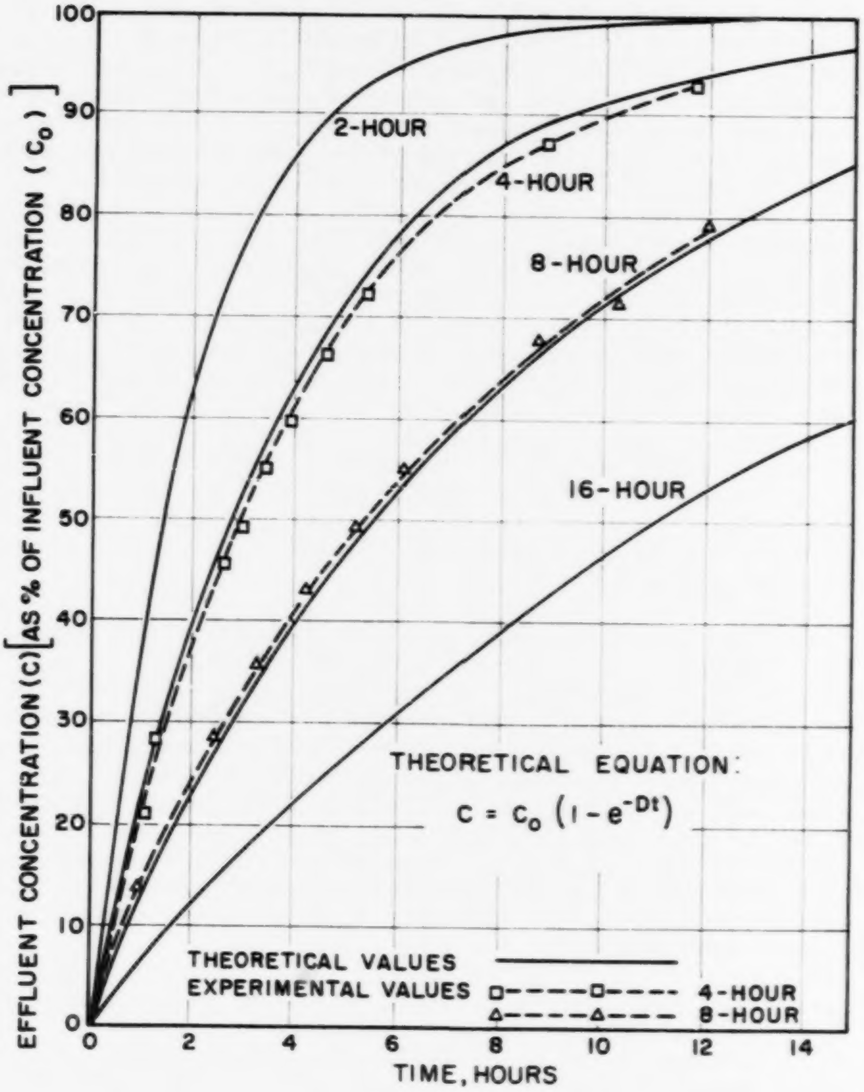

Figure 10. Comparison of theoretical and experimental dilute-in curves for a completely mixed laboratory reactor operated at various detention times (41).

Growth in the "Steady State"

Growth under continuous flow conditions may be studied in a reaction vessel fulfilling the requirements of complete mixing as outlined above, i.e., growth in a reactor such as shown in Figure 8. To start the system, one may fill the reactor with waste, add seed from a source such as sewage, and allow the cells to grow under batch conditions. The pumps may be started approximately when the solids have reached the maximum level (i.e., at the end of the substrate removal phase) or before. There are a number of ways to initiate continuous flow operation. The essential fact is that the system will, after a time, attain a condition (provided F is not exceedingly high or low) wherein values for X and S will remain essentially constant. The concentrations attained will be close to those exhibited in batch systems at the end of the substrate removal phase in instances in which S_0 in the batch reactor was equal to S_i in the continuous flow reactor. However, unlike the batch system (Figure 1), the biological solids concentration, X, would not decrease below this point; it would remain relatively steady, as would S. A continual feed stream allows the cells displaced from the reactor (by the factor F/V or D) to be replenished and X remains approximately constant. The substrate concentration remains approximately constant because cell yield, Y, remains approximately constant. In general, then, the system "approaches" a steady state with respect to X and S.

It is not necessary to ascribe a physiological significance to the term "steady state" since, like exponential growth, it may be defined on a strictly mathematical basis; i.e., $dX/dt = 0$, and $dS/dt = 0$. It has been experimentally demonstrated for pure cultures, where there is no chance for change in the predominating species except by mutation and, therefore, no change in the biological constants, that relative steadiness in X and S can be attained. Also, after considerable expenditure of experimental effort (22), it has been shown that a "pseudo" steadiness in X and S can be observed with heterogeneous populations of microorganisms, where there is much chance for changes in the predominating species, and therefore opportunity for change in the biological constants.

Having established the possibility of steady state growth experimentally, we may consider the mathematics of such growth. In Chapter IV it was stated that equation 14 describes the behavior of X in an exponential phase of growth characterized by a constant proportionality factor, μ, the specific growth rate. We can determine whether, in the steady state continuous growth situation, the cells are in the exponential phase by removing this constraint from μ as we have defined it and by assuming that the specific growth rate, μ, is $dX/dt)(1/X)$ in any phase of growth, exponential or otherwise. Considering X in the reactor, there is a continual pressure for biological solids to increase due to growth ($dX/dt = \mu X$) and to decrease due to cells being diluted out of the reactor ($dX/dt = -DX$). Since the total weight of cells is the volume multiplied by the concentration, the change in mass can be expressed in terms of a mass balance;

$$\begin{array}{lclcl}\text{Rate of change} & & \text{Rate of} & & \text{Rate of} \\ \text{in biological} & = & \text{increase due} & - & \text{decrease due} \\ \text{solids} & & \text{to growth} & & \text{to outflow}\end{array}$$

$$V\frac{dX}{dt} = V\mu X - FX \tag{25}$$

Since $\frac{F}{V} = D$, dividing through by V and substituting D yields:

$$\frac{dX}{dt} = \mu X - DX \tag{26}$$

However, we know from experimental observation that a state is attained wherein $dX/dt = 0$, or at least approximately so. Thus,

$$\mu = D \tag{27}$$

The identity above is a very important one. It states that the specific growth rate is equal to the dilution rate, D, i.e., F/V; therefore, it is subject to hydraulic control. The incoming flow rate to the reactor is then a very important physical parameter which can exert a considerable effect on the biological system. This fact has obvious engineering significance, since F is a system parameter which is more subject to engineering control than are the biological parameters μ_{max}, K_s, and Y.

Equation 27 has another significance which may not be generally appreciated. It states in essence that the specific growth rate, μ, is in fact a proportionality factor during exponential growth, and that completely mixed systems operating under conditions approaching a steady state with respect to X (i.e., $dX/dt = 0$) are operating in an exponential phase of growth. This conclusion follows from the fact that D was held constant; therefore μ is constant, and if μ is constant, by definition, growth is exponential. The biological solids concentration could not be held steady ($dX/dt = 0$) unless the cells were replenished at a constant specific rate, $\mu = D = \ln 2/t_d$. Again, this condition satisfies the previous definition of exponential growth, and if a once-through system such as that shown in Figure 8 is said to be in steady state ($dX/dt = 0$), it must also be said that exponential growth obtains.

It is now possible to write a mass balance for the reactor with respect to change in substrate concentration, S, and then to let dS/dt approach zero in the steady state:

Rate of change in substrate	=	Rate of increase due to inflow	-	Rate of decrease due to outflow	-	Rate of decrease due to consumption in reactor

$$V \frac{dS}{dt} = FS_i - FS - V \frac{\mu X}{Y} \tag{28}$$

As with the balance for biological solids, the rate of change in total mass of substrate, $(V)(dS/dt)$, in the reactor is the sum of the factors causing an increase and a decrease. The term for substrate consumption, i.e., $\mu X/Y$, comes directly from the equation

$$dS/dt = \frac{1}{Y}(dX/dt) \tag{21}$$

and the equation

$$\frac{dX}{dt} = \mu X \tag{14}$$

As in the case of the mass balance for biological solids, we can simplify the substrate balance by dividing by V and substituting D for F/V:

$$\frac{dS}{dt} = DS_i - DS - \frac{\mu X}{Y} \tag{29}$$

Also, since $\mu = D$, the dilution rate can be substituted in the consumption term, and letting $dS/dt \doteq 0$ as steady state is approached provides an equation for the average steady state value of X (i.e., $\bar{X}$), which can be written in terms of the average steady state value of S (i.e., $\bar{S}$)

$$\bar{X} = Y(S_i - \bar{S}) \tag{30}$$

This equation for the steady state value of X indicates that the solids concentration is dependent on the biological constant, Y, and on ΔCOD, wherein $\Delta\text{COD} = S_i - \bar{S}$. This is very similar to the expression for X in a batch system.

The equation for $\bar{X}$ is not really useful as a predictor for $\bar{X}$ unless $\bar{S}$ is known or can be predicted by an independent relationship not containing $\bar{X}$. Again, equation 27 ($\mu = D$) provides a useful relationship. Combining equations 27 and 18 (the Monod equation), but using the steady state concentration of S, i.e., $\bar{S}$, instead of S_o, we can write

$$D = \mu_{max} \frac{\bar{S}}{K_s + \bar{S}} \qquad (31)$$

from which

$$\bar{S} = K_s \frac{D}{\mu_{max} - D} \qquad (32)$$

an equation for $\bar{S}$ in terms of the biological system constants, K_s and μ_{max}, and the selected value of D (i.e., F/V) which is subject to engineering control. It is seen by equation 32 that the effluent substrate concentration, $\bar{S}$, is dependent, not on X, but on D and the biological constants, in the simple once-through (no cell recycle) system shown in Figure 8.

Again, it is emphasized that equations for $\bar{X}$ and for $\bar{S}$ can be shown experimentally to provide a reasonable assessment of the biological solids and substrate concentrations exiting the completely mixed reactor. These equations were presented as theory as early as 1950 by Monod (30). They were further developed and tested by Herbert, et al. (19) [also see Herbert (31)]. In all of these cases, the equations evolved from the assumption that very small changes in S caused essentially instantaneous changes in μ; i.e., that S in the Monod equation (equation 18) was not S_o in batch or $\bar{S}$, the steady state substrate concentration in a completely mixed continuous flow reactor, but S at any time and place. It can easily be shown that μ is actually not immediately sensitive to small changes in S (20, 21). For example, in a batch system, a considerable amount of substrate is removed during the time μ is constant, i.e., during exponential growth. This happens at initial substrate concentrations below those which will allow development of μ_{max} but it is readily observable only in systems for which K_s is not extremely small. Therefore, when equation 18 is written using S, rather than S_o, to describe growth in a batch system, the equation is not really correct. Our studies with both pure cultures and a heterogeneous population have shown that μ and S are not tightly coupled and that "slippage" in the equation may amount to 50% (20, 21). The workers cited above observed very low values of K_s in their studies with pure cultures, and this contributed to a theoretical development based on some incorrect assumptions which led to some misleading interpretations by subsequent researchers. Thus, there is an obvious inadequacy in the "theoretical" basis of equations 30 and 32. For this reason we have derived them as approximate empirical models for depicting "steady state" values of $\bar{X}$ and $\bar{S}$. In brief, we have taken a practical approach which should be more appreciated by engineers than by biological kinetics researchers. Experimentally, a relatively steady state develops, and these equations can be used to model it and to predict levels of $\bar{X}$ and $\bar{S}$; the steady state does not inevitably develop because of these equations or because of any theory on which they are based. The fact that a process can be described quantitatively with formulas of predictive value is not necessarily ascribable to correctness of theory or hypothesis. The equations are best looked upon as an outcome of enlightened empiricism. They are important equations because they are basic to description of the behavior in $\bar{X}$ and $\bar{S}$ of completely mixed, continuous culture reactors.

For further discussion of once-through reactors, demonstration of the pseudo-steadiness in X and S which develops using heterogeneous populations of sewage origin, and levels of $\bar{X}$ and $\bar{S}$ at various dilution rates, the reader is referred to an article by Gaudy, Ramanathan, and Rao in the research literature (22). Figure 11 is reproduced from this publication and shows the typical degree of steadiness developed in the laboratory reactor. It is apparent that S approaches a steady state value much more closely than does X. Using equations 30 and 32, it is possible to select values for μ_{max}, K_s, and Y (e.g., 0.5 hr^{-1}, 75 mg/l, and 0.6, respectively) for any value of S_i (e.g., 1000 mg/l) and to calculate values for $\bar{X}$ and $\bar{S}$ at various dilution rates (beginning with 1/24 and progressing toward $D = \mu_{max}$). These calculations have been made and will be shown in Figure 13 in the following section. On a plot of $\bar{X}$ and $\bar{S}$ vs. D it will be seen that $\bar{X}$ and $\bar{S}$ remain about the same over a fairly wide spread of D values (or detention times, $D = 1/\bar{t}$); the substrate concentration will remain rather low and will sweep upward sharply as D approaches μ_{max}. The biological solids concentration will fall rapidly (wash out). Comparison of such calculated values with those observed for heterogeneous populations in laboratory reactors validates use of these equations for once-through continuous flow systems (22).

Cell Recycle with Constant Concentration Ratio

If cells (sludge) are recycled to the reactor after being separated from the mixed liquor, as is the general practice with activated sludge processes (see Figure 12), there are, in addition to the engineering control possible through control of F and V, i.e., D, two other important parameters which are subject to engineering control. These are the hydraulic rate or ratio of sludge recycle and the concentration or concentration ratio of biological solids in the recycle flow. Mass balance equations for steady state operation of such a system have, as with the once-through reactor, been developed by workers in the field of microbial kinetics for pure cultures and, as with the once-through system, much research effort in our laboratories has been expended to determine their applicability to systems in which heterogeneous populations are employed.

The kinetic model to be discussed first is one which has been developed by Herbert (31). This model is useful, but we have not found it well adapted for use with activated sludge systems, and will present

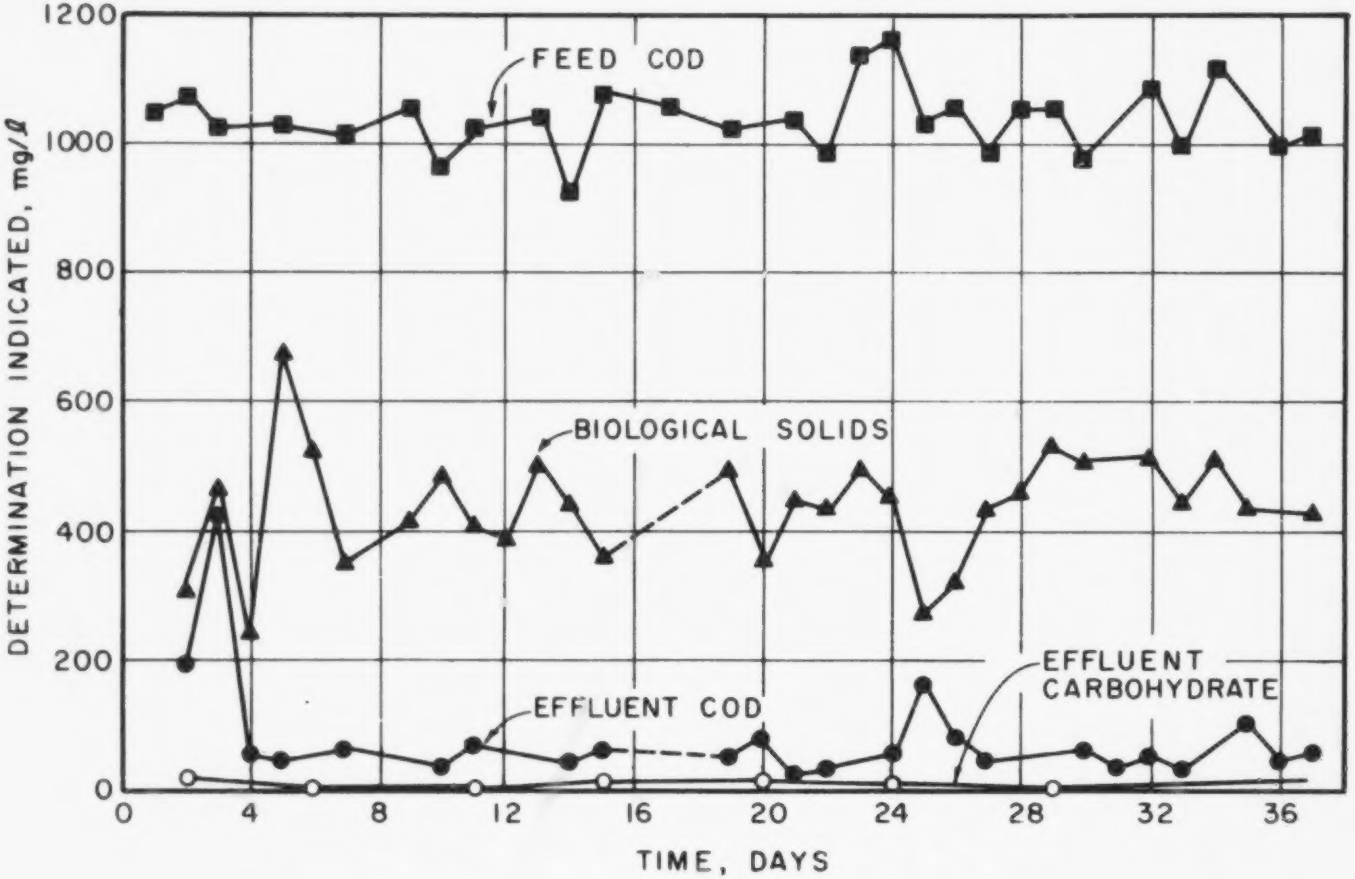

Figure 11. "Steady state" parameters measured in a completely mixed laboratory reactor of the once-through type, operated at a dilution rate of 0.33 hr^{-1} with nominal influent substrate concentration, S_i, of 1060 mg/l glucose COD (22).

a modification of it. First, it is important to show the equations of Herbert, since they provide a direct and simple expression of kinetic behavior in systems employing cell recycle.

Figure 12 shows a flow sheet for a reactor with recycle. After leaving the reactor, the mixed liquor is passed through a concentration step (clarifier, centrifuge, etc.); in Herbert's case, a centrifuge was used. Cells are concentrated to the desired degree for recycle. The parameter which is employed as a systems constant in Herbert's model is the sludge concentration factor, c, which is the ratio of cell concentration, X_R, in the recycle to cell concentration, X, in the reactor. The parameter for recycle flow is termed the recycle ratio, α, which is the ratio of recycle flow rate to the rate of flow, F, of the incoming medium. Exiting the system is the flow, F, containing the excess or waste cell concentration, X_e. The dilution rate, D, for the system as a whole is as before, F/V, but the dilution rate for the reactor, D_r, is now increased (detention time decreased) because of the recycle flow and is equal to $(1 + \alpha)D$.

The two new system controls (constants), α and c, effect changes in the mass balance for the reactor. The materials balance for the rate of change of biological solids in the reactor is:

$$\text{Rate of change in biological solids} = \text{recycle rate} + \text{growth rate} - \text{outflow rate}$$

$$V \frac{dX}{dt} = \alpha F c X + V \mu X - F(1 + \alpha)X \qquad (33)$$

The materials balance for rate of change of substrate in the reactor is:

$$\text{Rate of change in substrate} = \text{inflow rate} + \text{recycle rate} - \text{outflow rate} - \text{consumption rate}$$

$$V \frac{dS}{dt} = FS_i + \alpha FS - F(1 + \alpha)S - V \frac{\mu X}{Y} \qquad (34)$$

Dividing both equations by V, substituting D for F/V, letting dX/dt and dS/dt equal zero (i.e., steady state conditions) and substituting the Monod relationship for $\bar{S}$, equations 33 and 34 can be solved simultaneously, yielding the following equations for steady state concentration of biological solids, $\bar{X}$, and substrate concentration $\bar{S}$:

$$\bar{X} = \frac{Y}{(1 + \alpha - \alpha c)} (S_i - \bar{S}) \qquad (35)$$

$$\bar{S} = \frac{K_s D(1 + \alpha - \alpha c)}{\mu_{max} - D(1 + \alpha - \alpha c)} \qquad (36)$$

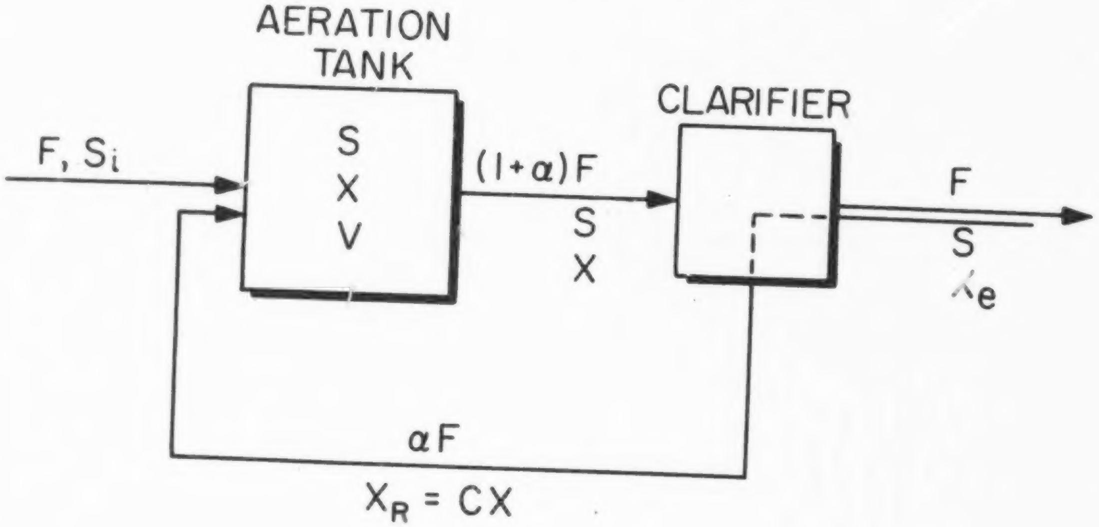
AERATION TANK
S
X
V
F, S_i
$(1+\alpha)F$
S
X
CLARIFIER
F
S
X_e
αF
$X_R = CX$

Figure 12. Flow diagram for a continuous flow, completely mixed reactor employing cell recycle at a constant ratio, c, of recycle solids, X_R, to aeration solids, $\bar{X}$. Diagram represents the kinetic model proposed by Herbert (31).

Letting the factor $(1 + \alpha - \alpha c) = A$

$$\bar{X} = \frac{Y}{A}(S_i - \bar{S}) \tag{37}$$

$$\bar{S} = \frac{K_s AD}{\mu_{max} - AD} \tag{38}$$

Equations 37 and 38 are those developed by Herbert (31). They provide a fairly accurate description of the steady state levels of biological solids and substrate concentrations for completely mixed continuous culture devices employing cell recycle; they help delineate important kinetic principles, and extend those developed in arriving at equations 30 and 32 for once-through systems. Selecting the same values for the biological constants as those used in making calculations for the once-through system and selecting $\alpha = 0.25$ and $c = 4.0$, the calculated levels of $\bar{X}$ and $\bar{S}$ for $S_i = 1000$ mg/l are plotted for various values of D in Figure 13. The values of $\bar{X}$ and $\bar{S}$ for the once-through system of Figure 8 are plotted in broken lines on the same figure to illustrate the effect of recycling solids. The values of 0.25 for α and 4.0 for c were chosen because they are values which are close to those typical of many field installations. Design hydraulic recycle flow rates of 0.25 times the design flow are commonly used, and it is not uncommon to observe recycle sludge concentrations at least four times the concentration of biological solids in the aeration tank. Thus, the values chosen are realistic ones (as are those for the biological constants). The term A in equations 37 and 38 permits selection of a wide combination of values of c and α. It is obvious that selection should be such that A does not go to zero. For example, one could not select $\alpha = 0.25$ and $c = 5$. If one were to concentrate the return solids much beyond $c = 4$, α could be reduced.

As seen from the figure, cell recycle permits the existence in the reactor of considerably higher concentrations of cells, X, than is possible in a once-through reactor, and consequently allows greater utilization of substrate at lower detention times (higher dilution rates). In both cases, the biological constants are the same, but with cell recycle the engineering or physical constants, α and c, exercise a control over the system over and above that in a once-through system which seeks the natural steady state substrate and solids levels governed solely by the biological constants, Y, K_s, and μ_{max}, and the dilution rate, D. It was shown previously that for a once-through reactor, $\mu = D$ (equation 27). However, for a system employing cell recycle, μ is less than D. According to Herbert's derivation of the equations for this kinetic model (31), μ is related to D by a factor depending upon the values chosen for α and c, i.e., $\mu = AD$. From this it follows that the specific growth rate, μ, in a system employing cell recycle can be considerably lower than that in a once-through system

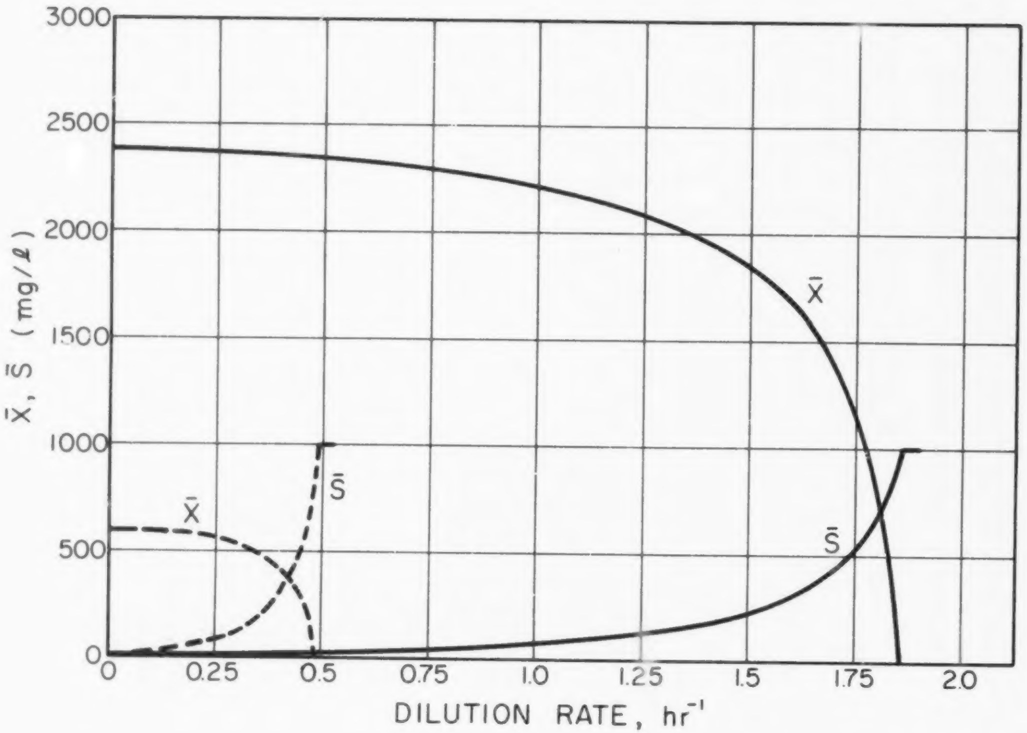

Figure 13. Comparison of predicted levels of $\bar{X}$ and $\bar{S}$ at various dilution rates in a once-through reactor (broken lines) and a reactor employing cell recycle concentration ratio, c, as a system constant in accordance with the kinetic model equations of Herbert. Values used for calculation were: μ_{max}, 0.5 hr^{-1}, K_s, 75 mg/l; Y, 0.60; α, 0.25; c, 4.0, S_i = 1000 mg/l.

with the same system dilution rate. For example, in the system shown in Figure 13 where $1 + \alpha - \alpha c = 0.25$, $\mu = 0.25D$. Thus, at a D of 0.2, μ is only 0.05 hr^{-1} rather than 0.2 hr^{-1} as in the once-through system. Therefore, on the average, the probability that a cell will divide during a pass through the reactor with recycle is much less than in the once-through system. However, as pointed out previously, according to the mathematical definition of exponential growth (μ is constant), if A and D are kept constant and if $dX/dt = 0$, growth is exponential even at the very slow specific growth rates of recycle systems.

We have in Herbert's model, a simple and direct, albeit empirical, picturization of completely mixed continuous microbial growth systems in steady state conditions. One could also use these model equations in $\bar{X}$ and $\bar{S}$ to assess the effect on effluent substrate levels of various values of the biological constants as was done for values of D in Figure 13. It was shown in Figure 13 that $\bar{S}$ remains fairly low over a wide range of dilution rates; i.e., that these systems are capable of delivering rather low effluent concentrations of $\bar{S}$ at high dilution rates. However, operation at very low detention times cannot now be recommended because of problems with sludge flocculation and settleability at high specific growth rates, and because in this area even more than in other engineering areas of functional design and operation, safety factors are needed. In any event, it can be seen through study of these model equations and in charting the behavior of $\bar{X}$ and $\bar{S}$ that biological treatment systems of this type possess potential capabilities for removing substrate loadings much in excess of present day loadings.

One cannot expect the experimental data (lab or field) to trace a perfect fit to such a model for the simple reason that for heterogeneous populations the biological constants vary somewhat even if one could, as in the lab, operate with S_i, D, α, and c steady. Herein lies an important point, or engineering concept, which deserves emphasis. One could use these equations in design with the assumption of constant values of S_i, D, α, and c, in accordance with the model. But if the design provided no means for attempting to hold these physical (engineering) parameters constant, the operator could not hope to attain the designer's goal. In brief, if designers use a model, the design must attempt to accommodate the model; this is a major requirement of "engineering control" of processes. An equalization or surge basin could attenuate swings in S_i and D. If it were aerated, it could also provide some degree of treatment as an unsteady once-through reactor. The recycle flow is fairly easily controlled. Control of c requires sensing for X in both the reactor mixed liquor and the clarifier underflow, and a means of adjusting X_R to hold c approximately constant at the chosen value of X_R/X. This latter requirement is not very easily attained, especially if X is subject to much variation. As with the once-through model, the recycle model was originally proposed on the assumption that pure cultures would be employed, for which K_s, μ_{max}, and Y are constants, and we have undertaken experimental studies to determine the applicability of this recycle model for heterogeneous (natural) populations.

Experimentation with the once-through model had indicated some variation in S and somewhat more variation in X (e.g., Figure 11), but average values of S and X during operation at various values of D did produce dilute-out patterns in S and X in general accord with equations 30 and 32 (22). To test the applicability of Herbert's recycle model to heterogeneous populations, a laboratory pilot plant was set up in accordance with the flow diagram for the model which was shown in Figure 12 (27). The biological sludge was not recirculated directly from the bottom of the clarifier, but was channelled to another tank in which the recycle sludge concentration, X_R, was adjusted in accordance with $c = X_R/X$. Data obtained for one set of values of the operational constants are shown in Figure 14 (taken from reference 27). The concentration factor, c, was purposely held low (1.5) in these experiments in order to ensure that we could always obtain the required X_R by diluting underflow. The sludge recirculation ratio, α, was maintained at 0.25. Figure 14 shows the variation in X and X_R for S_i of 1000 mg/l glucose (1060 mg/l COD)(top portion) and in substrate, S, in terms of COD and carbohydrate (lower graph). In the example shown, the detention time in the activated sludge tank was three hours; i.e., $D_r = 1/3\ hr^{-1}$, and the overall dilution rate, D, was $1/3.75\ hr^{-1}$ i.e., $D_r/(1 + \alpha)$. It is seen that in general, c was maintained close to the selected 1.5 ratio. Also, it is apparent that X varied; the reactor did not attain a "steady state." The values for S (COD) also varied considerably, but the scale is somewhat expanded (8 to 1, as compared with the top graphs), and the effluent substrate concentration, S, could be adjudged as fairly steady over the two-month period of operation at this detention time. The values of S measured as carbohydrate (using the anthrone test) showed only slight variation. Both the COD and carbohydrate measures for S shown in the figure were obtained by analysis of the membrane filtrate. However, it should be noted that during this run, had COD been plotted for the clarifier supernatant (actual effluent), the values would be only slightly higher because the sludge settling characteristics were rather good. In general, the supernatant COD was 25-30 mg/l higher than the filtrate COD. The average value of aeration solids during this period was 753 mg/l, and the average effluent COD was 79 mg/l. These values along with other parameters for operation at this and other values of D are given in Table III. In Figure 15, the average experimental values for X and S from Table III are plotted as the "steady state" levels ($\bar{X}$ and $\bar{S}$) for the four dilution rates examined. The heavy lines through these experimental points describe a portion of a dilute-out in X and rise in S as D increases. This experimental behavior agrees with the dilute-out pattern which is the expected behavior according to equations 37 and 38. The dotted curves labelled X_1 and S_1 show the predicted levels of $\bar{X}$ and $\bar{S}$ calculated from equations 37 and 38 using values of the biological constants, μ_{max}, K_s, and Y determined for the cells present at each dilution rate. The μ_{max} and

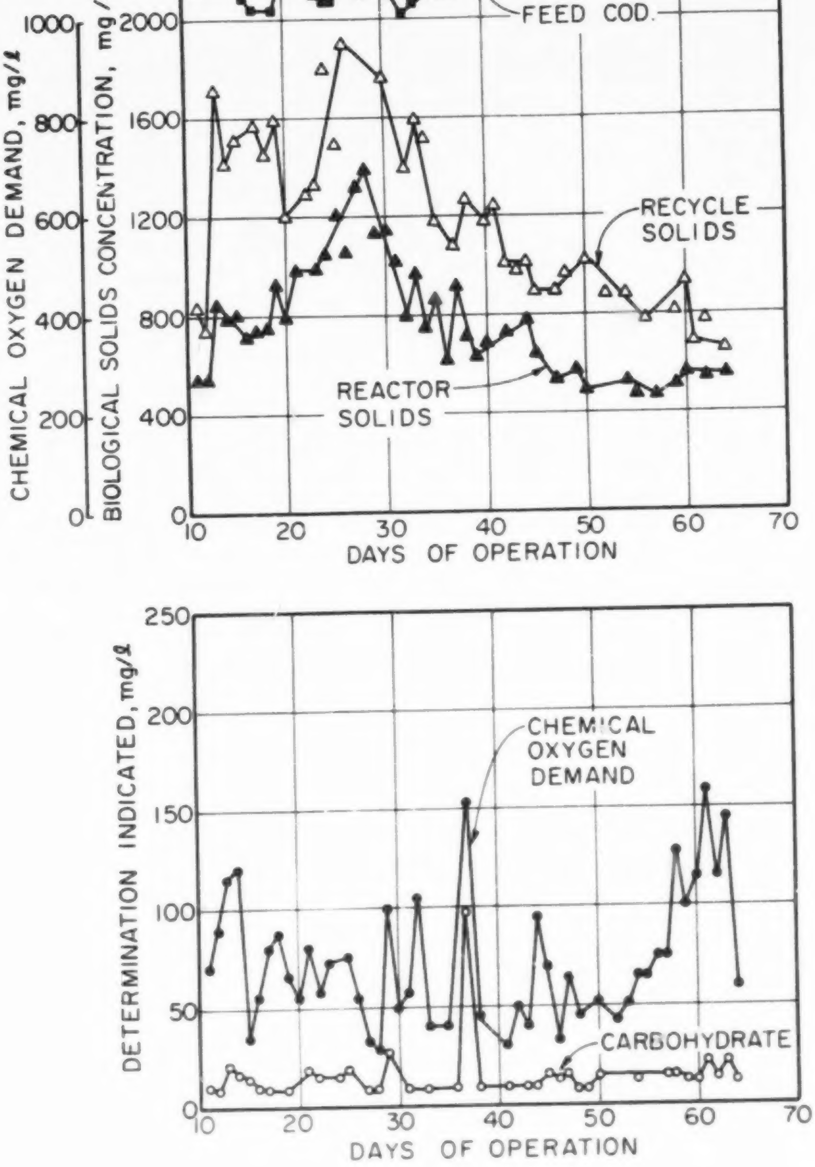

Figure 14. Operational data for a laboratory reactor operated according to the kinetic model of Herbert with cell recycle at a constant recycle ratio, c, of 1.5. S_i = 1060 mg/l glucose COD, α = 0.25, aerator detention time = 3 hr (27).

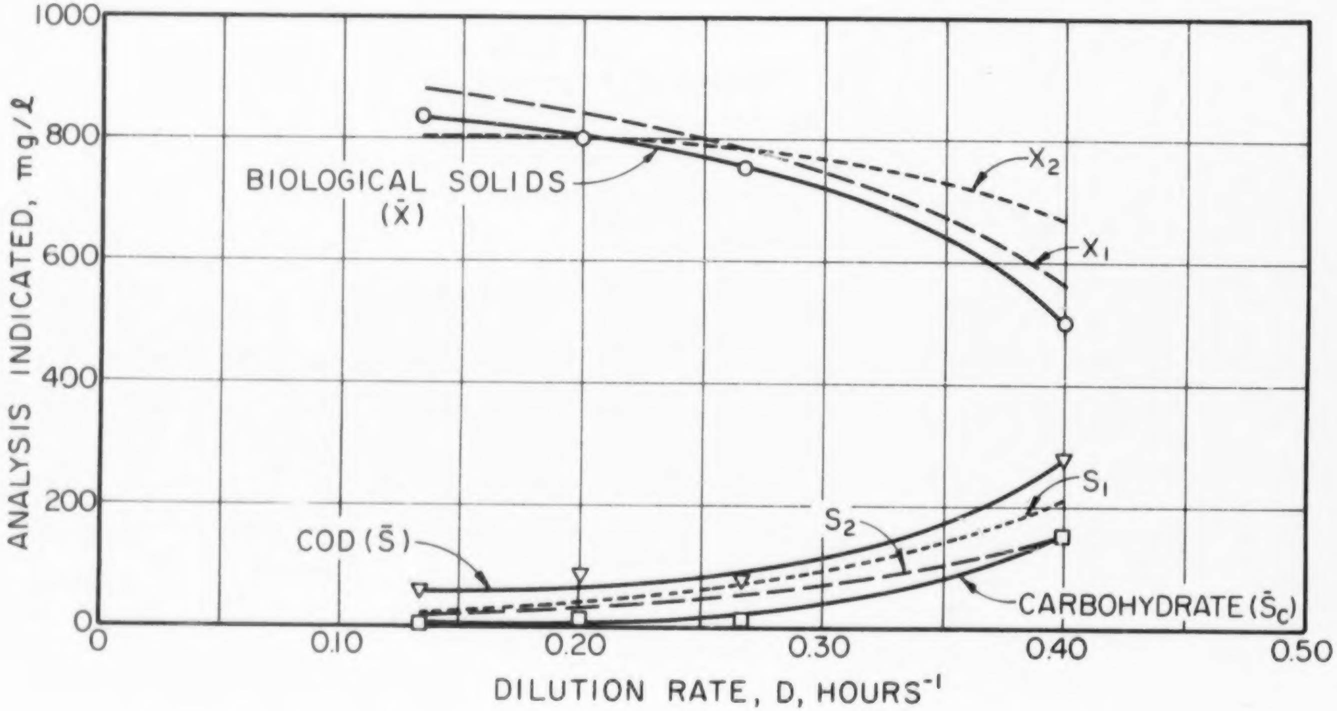

Figure 15. Comparison of observed (solid lines) values of $\bar{X}$ and $\bar{S}$ with values predicted (broken line curves) by the kinetic model equations of Herbert using experimentally determined values of the biological constants (see text for details)(27).

TABLE III

OPERATIONAL PARAMETERS FOR A COMPLETELY MIXED CONTINUOUS FLOW REACTOR WITH CELL RECYCLE

Dilution Rate, hr^{-1}	Mean Reactor Residence Time, $\bar{t}$, hr	$\bar{X}$[a] mg/l	$c\bar{X}$[b] mg/l	$\bar{S}$[c] mg/l	$\bar{S}_i$[d] mg/l	$\bar{S}_c$[e] mg/l	$\frac{(S_i-\bar{S})}{\bar{X}\bar{t}}$[f] mg COD/ mg SS/hr	Loading Factor lb COD/ lb SS/day	COD Removal Efficiency %
1/7.5	6	837	1295	60	1040	6	0.195	9.94	94.2
1/5	4	797	1236	87	1060	15	0.305	16.0	91.7
1/3.75	3	753	1121	79	1067	17	0.438	22.6	92.6
1/2.5	2	500	861	277	1065	156	0.788	51.1	74.0

a. $\bar{X}$ = mean biological solids
b. $c\bar{X}$ = mean biological solids in the recycle
c. $\bar{S}$ = mean effluent COD
d. $\bar{S}_i$ = mean influent COD
e. $\bar{S}_c$ = mean effluent carbohydrate
f. COD removal rate
$\alpha = 0.25$
$c = 1.5$
S_i = 1060 mg/l glucose COD

K_s values were obtained by taking a sample of sludge as seed material at various times during the operational period and using these samples for batch growth studies run as previously described. The value for Y was calculated directly from the pilot plant data. The curves labelled X_2 and S_2 were calculated using average values of μ_{max}, K_s, and Y for all the dilution rates examined. In general, it can be concluded that the model does describe (predict) fairly well the behavior of these systems in which the microbial population was not a pure culture but a heterogeneous bio-mass, i.e., an activated sludge.

After a large expenditure of research effort we are thus able to conclude that the model proposed by Herbert does describe the observed biological behavior. It is appropriate to examine the model, and to defend and criticize it with regard to the reasonableness of the approach before going further. It is first and most importantly a simple one unencumbered by a multitude of biological constants which need to be determined; only μ_{max}, K_s, and Y are required. It may be argued that in setting up the materials balance there was no term for substrate consumption for maintenance of existing solids, i.e., substrate was considered as being consumed only for respiration and growth of the bio-mass. The amount of exogenous carbon source used by a microbial cell simply to maintain itself intact is very, very small compared to the amount needed for growth. It is so small that it is not a significant factor and, indeed, is very difficult to measure except for extremely high cell concentrations. Also, in the mass balance for X, no term was inserted for a decrease in X due to "endogenous" degradation of the bio-mass. This omission is also defensible. In the presence of exogenous substrate (S_i coming into the reactor continually), the net effect is an increase in the sludge concentration. If this were not so, there would be no excess sludge to dispose of. (The special situation in which endogenous metabolism, i.e., autodigestion, is extremely important, the extended aeration process, will be discussed in Chapter VIII.) At any rate, it is impossible to determine whether, or how much, endogenous metabolism occurs in the presence of exogenous substrate, and non-measurable quantities have no practical value. Since net increase in solids is measurable, endogenous metabolism is automatically accounted for if it does occur, and there is really no good reason for complicating general, working-model equations for activated sludge by insertion of a term for endogenous degradation. The question posed is: Why overcomplicate for the sake of not missing anything which could possibly affect X and S? There seems to be no justifiable reason for doing so. One can surely be cognizant of the fact that these phenomena can occur, but if they do not play a significant role (maintenance) or are not measurable as separate reactions (endogenous metabolism), they can and should be omitted from the general model.

Thus, it can be seen that the model presented could be used as an approach to plant design, an approach which is in accord with biological principles relating the major parameters which govern the function (operation) of the plant. Equalization tanks could help even out fluctuations in D and Si; α can be controlled fairly easily, but a constant sludge concentration factor causes difficulty in operation. It was observed in once-through systems (e.g., Figure 11) that while S attained a relatively steady level, a steady state with respect to X was only roughly approached. When sludge recycle is practiced (e.g., Figure 14), X is subject to great fluctuation. If one operates in accordance with the model, it is required that c remain constant. In order to accomplish this, one is required to change the recycle sludge concentration, X_R, when he detects a change in X in the reactor. This mode of operation requires rather close monitoring of the plant (which should be done in any event). A more serious defect of this model results from solids variations due to heterogeneity of the population, so that operation at a constant c actually militates against steadiness in X. The operator may take a sample of mixed liquor and find that since the last sample (e.g., the previous day or shift), the aeration solids concentration has increased. To hold c constant, he then must increase X_R. This will not help to steady X, but will tend to cause a further increase in X. Thus, while the model may work well with pure cultures and while it offers an excellent approach to kinetic picturization of the behavior of S and X for the activated sludge process and could be employed as a design tool, operation in accordance with this model is difficult, and militates against steadiness in X. One could let X vary, so long as S remained steady, since the prime concern is naturally with the effluent quality and with reliable delivery of as low an S as is economically possible. However, since constancy of c is the most difficult operational requirement of the model and militates against predictability of $\bar{X}$ in activated sludge systems (heterogeneous microbial populations), we asked the following question: What happens to the model if, instead of holding c as an operational constant, we hold X_R constant (an engineering expedient which we could facilitate in practice much more easily than constant c)?

Cell Recycle with Constant Return Solids Concentration

A flow sheet for the system operated with X_R constant is given in Figure 16. It is essentially the same as Figure 12, but an aerated sludge consistency and holding tank (aeration tank 2) is placed in the line. Exiting the process is the effluent substrate concentration S and excess sludge, X_e. The total flow leaving the system is F (part of the flow is the supernatant and part is the flow of waste sludge).

Assuming, as in previous calculations, that the biological solids in the incoming waste flow is negligible compared to solids in the reactor, the balance equation for the rate of change in reactor solids can be written as follows:

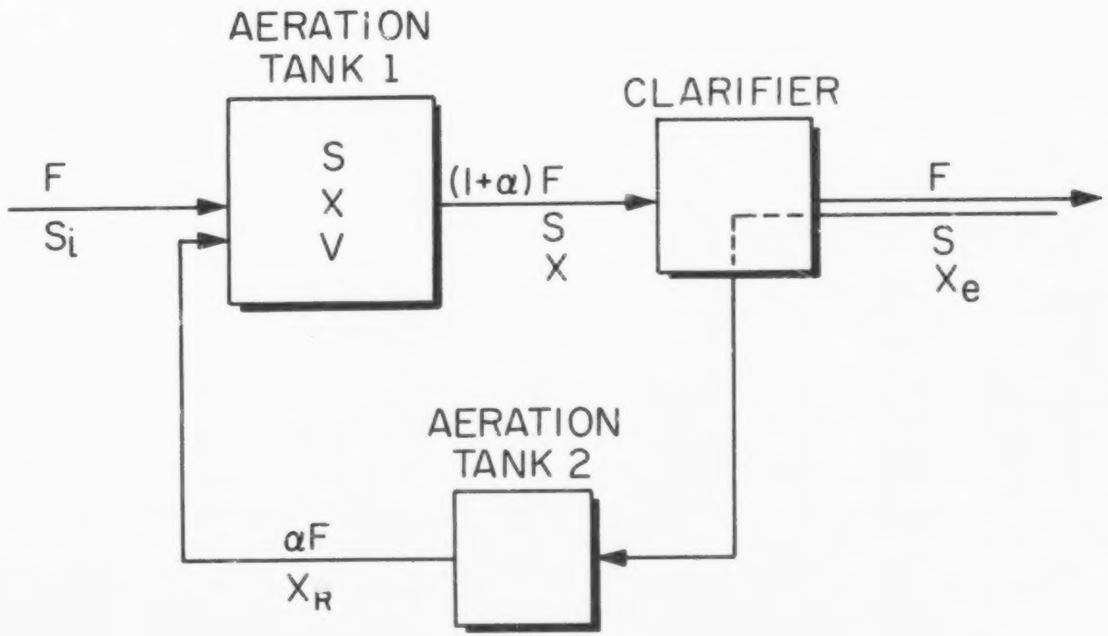
AERATiON
TANK 1
S
X
V
F
S_i
$(1+\alpha)F$
S
X
CLARIFIER
F
S
X_e
AERATION
TANK 2
αF
X_R

Figure 16. Flow diagram for a continuous flow, completely mixed reactor employing cell recycle at a constant recycle solids concentration, X_R. Diagram represents the kinetic model proposed by Ramanathan and Gaudy (32).

$$\begin{array}{ccccccc} & & \text{(recycle)} & & \text{(growth)} & & \text{(outflow)} \\ VdX/dt & = & \alpha FX_R & + & V\mu X & - & (1+\alpha)FX \end{array} \tag{39}$$

Proceeding as before, and setting dX/dt = 0:

$$\mu = D\ \left[(1+\alpha) - (\alpha X_R/\bar{X})\right] \tag{40}$$

The symbol X has now been replaced by the average steady state value $\bar{X}$, in accordance with attainment of steady state in X; i.e., dX/dt = 0.

The materials balance equation for the change in S through the reactor is:

$$\begin{array}{ccccccc} & & \text{(inflow)} & & \text{(consumption)} & & \text{(outflow)} \\ VdS/dt & = & FS_i & - & \mu XV/Y & - & (1+\alpha)FS \end{array} \tag{41}$$

from which

$$\bar{X} = (YD/\mu)\left[S_i - (1+\alpha)\bar{S}\right] \tag{42}$$

Substituting μ from equation 40 in equation 42, an expression for $\bar{X}$ in terms of $\bar{S}$ is obtained, and one for $\bar{S}$ in terms of $\bar{X}$ is obtained by substituting the Monod relationship for μ in terms of $\bar{S}$ in equation 40. Solution of these similtaneous equations leads to the quadratic form in which $\bar{X}$ and $\bar{S}$ are given as follows (for more detailed presentations of the intermediate steps in the derivation, see reference 32):

$$\bar{X} = \frac{Y\left[S_i - (1+\alpha)\bar{S}\right] + \alpha X_R}{(1+\alpha)} \tag{43}$$

$$\bar{S} = \frac{-b \pm \sqrt{b^2 - 4ac}}{2a} \tag{44}$$

where

$$a = \mu_{max} - (1+\alpha)D \tag{45}$$

$$b = D\left[S_i - (1+\alpha)K_s\right] - \frac{\mu_{max}}{(1+\alpha)}\left[S_i + \alpha X_R/Y\right] \tag{46}$$

$$c = K_s D S_i \tag{47}$$

These equations can be employed to depict the behavior of $\bar{X}$ and $\bar{S}$ for various values of μ_{max}, K_s, Y, S_i, D, α, and X_R. It is readily appreciated that the calculations can be facilitated by use of a computer. It is interesting to compare the values of $\bar{X}$ and $\bar{S}$ for this model with those for the recycle system shown in Figure 13. Such a comparison is presented in Figure 17. The curves labelled B are those representing Herbert's model and are replotted from Figure 13. The curves labelled A were computed from equations 43 through 47. The same values for all system parameters except recycle solids were used in calculating the two sets of plotting points. The only difference in the two systems is that c, i.e., X_R/X, is held constant at a value of 4.0 for curves B, while X_R is held constant at a value of 10,000 mg/l for curves A. The latter value for recycle solids concentration was chosen as one which is readily attained by quiescent settling. Two things are particularly important to note in this figure. First, operation at constant X_R flattens the dilute-out pattern of the system, i.e., provides more stability at high dilution rates; secondly, at dilution rates which would usually be considered for use in activated sludge systems (certainly those below 1.0), the values for $\bar{S}$ are the same for both models.

The model in which c is a system constant is more in keeping with justifiable kinetic theory, because it contains no assumption which prevents a system from diluting out. In the model we are recommending for activated sludge, total dilute-out is impossible (X_R is a constant), and X can reach only a lower level of X_R $(\alpha /1 + \alpha)$; also the assumption of zero substrate concentration in the recycle flow (see substrate balance equation 41) is unrealistic at extremely high values of D because S cannot rise to S_i but can only approach $S_i(1/1 + \alpha)$. However, at realistic values of D, the consequences of these assumptions are immaterial; thus the engineering expediency in making X_R a system (design) constant is entirely defensible and advisable. These points are developed more fully in reference 32.

The behavior of the kinetic model in $\bar{X}$ and $\bar{S}$ which we are herein recommending has been examined computationally for various values of μ_{max}, K_s, Y, α, X_R, and S_i at various values of D (32). All of these factors are important kinetic parameters which affect both $\bar{X}$ and $\bar{S}$. However, at a reasonably high value for S_i (e.g., 1000 mg/l), changes in the biological constants μ_{max}, K_s, and Y do not appear to exert as much effect on $\bar{X}$ and $\bar{S}$ as do the physical constants α and X_R (32). In general, selection of $\alpha = 0.25$, which is a value widely employed traditionally, appears to be a rather good one for X_R values which can be reasonably attained in practice (e.g., 10,000 mg/l). The values for these parameters which are in one case (i.e., α) nominally attained in the field, and in the other case (i.e., X_R) are usually attained fortuitously, are rather satisfactory for dilution

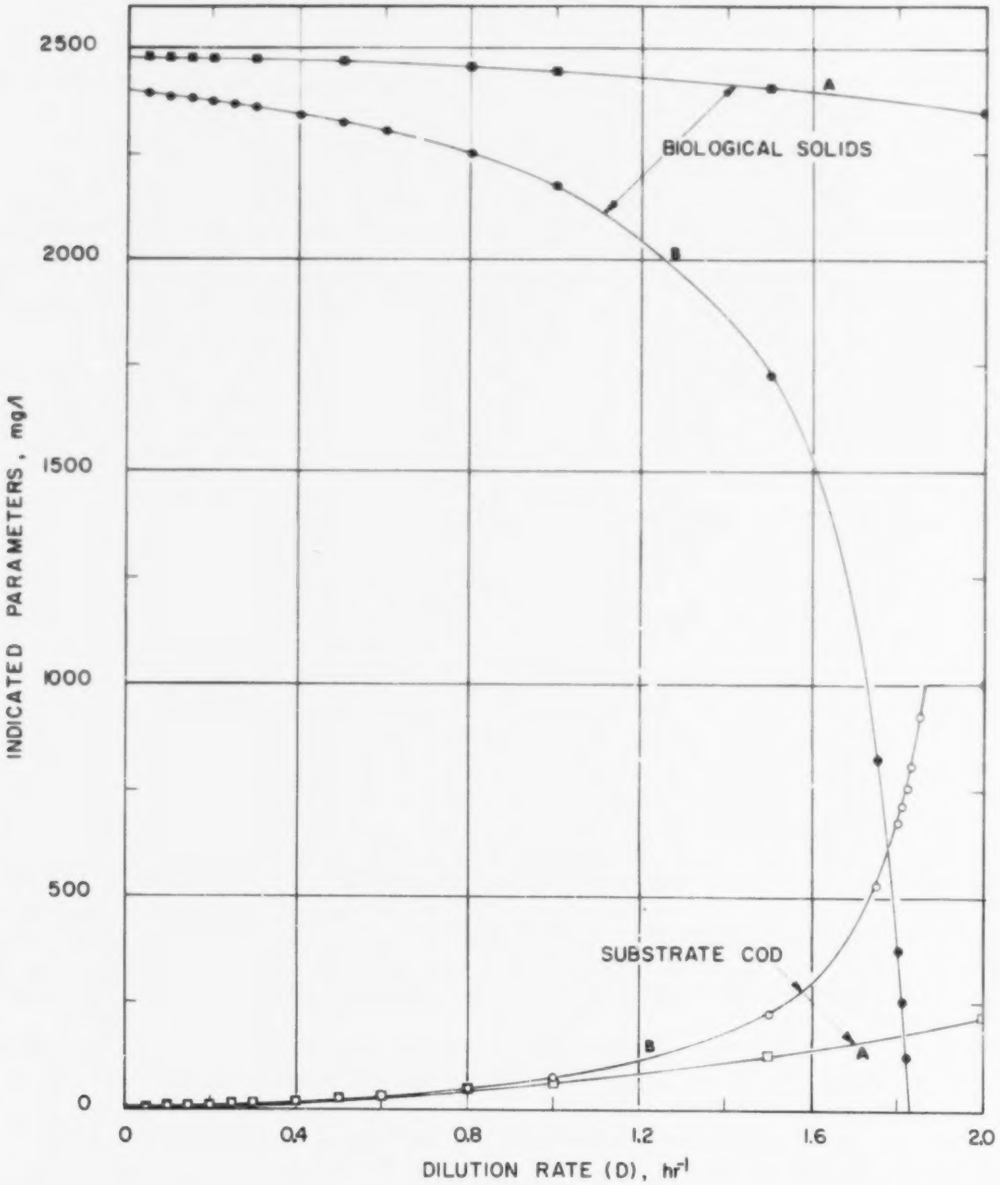

Figure 17. Comparison of values for $\bar{X}$ and $\bar{S}$ at various dilution rates predicted by the two kinetic models for cell recycle. Curves A are those predicted by the model equations of Ramanathan and Gaudy (32). Curves B are those predicted by the model equations of Herbert (31).

rates of less than 1/2 hr^{-1} and organic substrate loadings of 1000 mg/l and below.

In general, the model we have presented tends to explain why activated sludge plants have, on the basis of experience, gained a good reputation as substrate removers. Also, the model shows that unless one does control α, X_R, S_i, and D, they can give variable results in $\bar{S}$ (the most critical parameter). It is rather fortunate that the biological parameters, μ_{max}, K_s, and Y, which are not really subject to engineering control (unless predominance of species can be controlled at some point in the future), can be expected to exert less effect on performance than do the parameters which can be subject to engineering control (α, X_R, S_i, and D).

It is emphasized again that the operating engineer stands little chance of operating the system steadily and reliably unless the means for exerting control over these selectable variables are provided for by the design engineer. Conversely, if they are provided in the design, they are of little avail unless used by the operating engineer.

In Figure 18, the computed steady state levels of $\bar{S}$ and $\bar{X}$ at various dilution rates for S_i levels up to 5000 mg/l are plotted in accordance with our kinetic model (equations 43 through 47). The values employed for the biological constants as well as the engineering constants are those used previously (e.g., for calculating the curves shown in Figure 17), since they appear to represent values that are reasonably expectable on a biological basis and reasonably attainable from an engineering viewpoint, respectively. It can be seen that the efficiency of substrate removal remains fairly high over a wide range of feed substrate (S_i) concentrations, especially for dilution rates of 0.5 hr^{-1} and below. This dilution rate corresponds to a reactor dilution rate, D_r, of 0.625 hr^{-1}, i.e., a reactor detention time of 1.6 hours. At higher dilution rates, as S_i is increased, substrate leakage increases significantly. In Figure 19, the curves for D values of 0.1, 0.5, 1.0, and 2.0 hr^{-1} for S_i up to 1000 mg/l are plotted in expanded scale. Figure 18 was intended to show the overall behavior of the model equations, but the range of S_i up to 1000 mg/l is probably the one of most interest, since for many types of waste water the biologically available organic matter (measured as ΔCOD) is within this range.

A graph such as this could be employed as a design "guide," but its real value is that it shows the potential capability of the process. Estimates for design should be revised upward from this baseline guide. A constant balancing of "theory" vs. "engineering judgment" or "experience" is as necessary (or even more so) in this area as it is in any other area of engineering. For example, in Figure 19, the "chart" says that at a feed COD of 1000 mg/l, just over 60 mg/l COD ought to appear in the effluent at a D of 1.0 (reactor detention time of 0.8 hour). Thus well over 90% efficiency is predicted by the model at this low detention time. In determining whether the

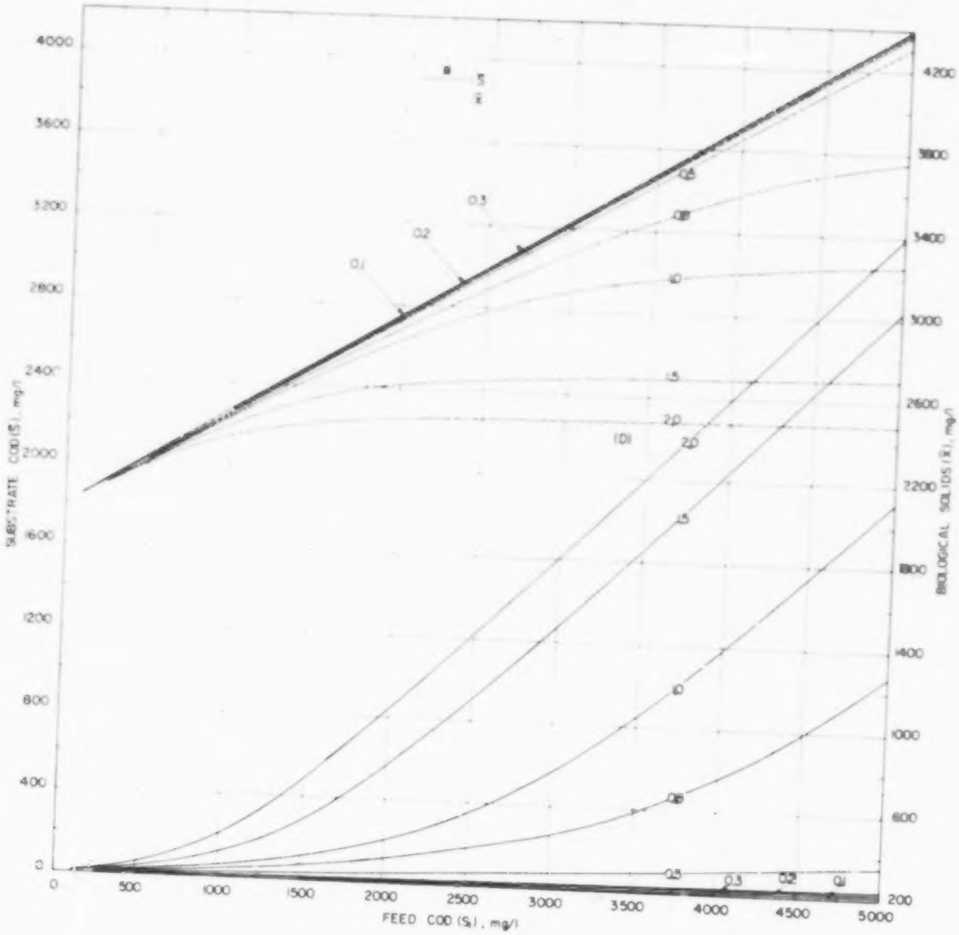

Figure 18. Predicted levels of aeration solids, $\bar{X}$, and effluent COD, $\bar{S}$, for a range of organic loadings, S_i, at various dilution rates. Curves were computed according to the model equations of Ramanathan and Gaudy (32) using the following values for system constants: μ_{max}, 0.5 hr^{-1}; K_s, 75 mg/l; Y, 0.60; α, 0.25; X_R, 10,000 mg/l.

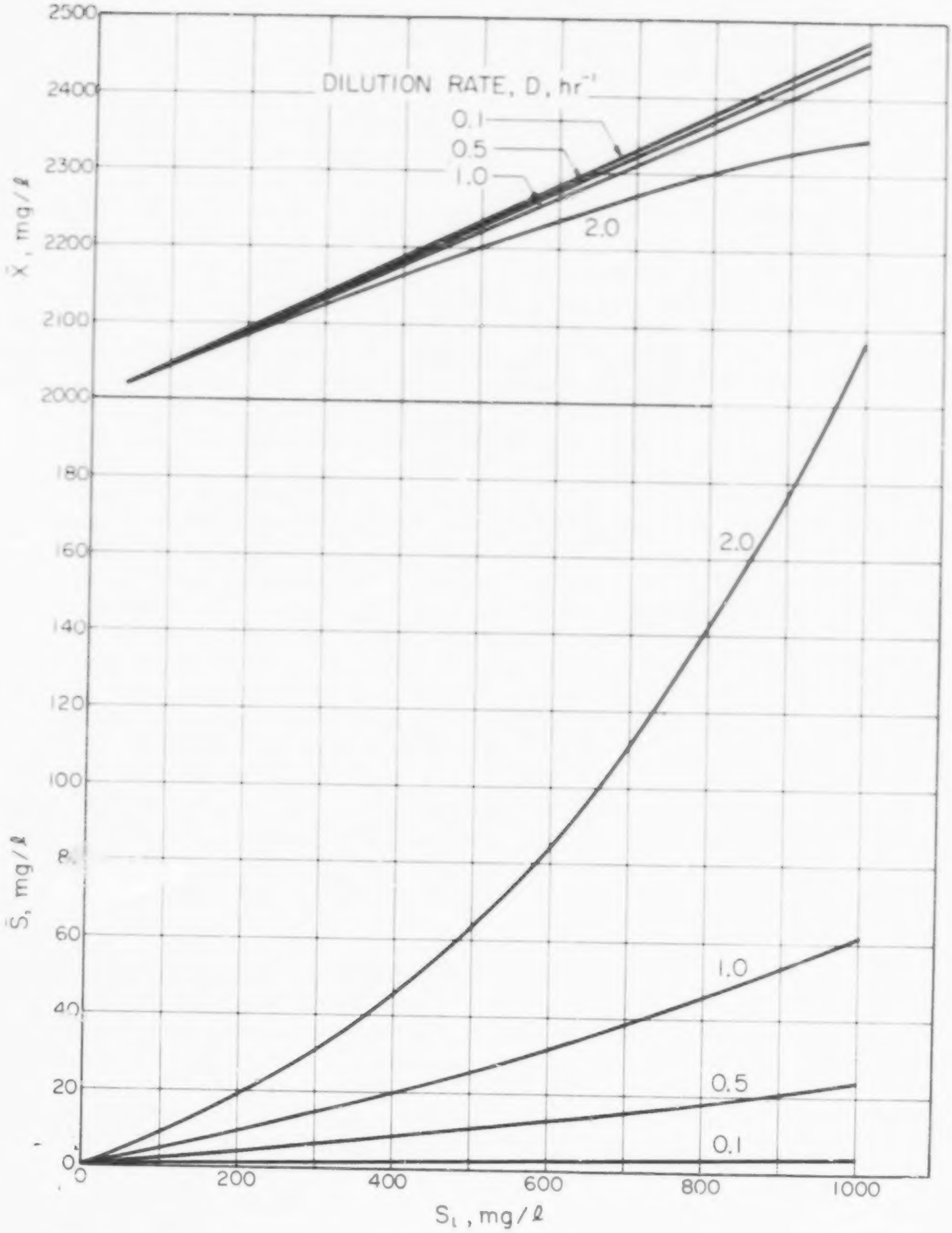

Figure 19. Expanded scale plot of the curves from Figure 18 for organic loadings up to 1000 mg/l COD.

prediction is reasonable, e.g., in checking performance in treatability studies, one needs to consider first that the substrate COD of the chart is just that: not total COD, but the portion of organic matter in the waste which is capable of being utilized as carbon source by an acclimated heterogeneous microbial population, i.e., the parameter ΔCOD developed in Chapter II. For example, a waste containing biologically resistant organic matter may exhibit a very large residual COD even after the biologically usable (i.e., biologically oxygen-demanding) organic matter has been removed. Thus one could not expect to register COD values as low as shown, even when these values, on the basis of utilizable carbon source, were attained. Use of a test for residual biologically degradable material in the effluent, as recommended in Chapter III would, however, demonstrate the high efficiency of the process. It should also be realized that CODs run on potable tap water can sometimes be registered as high as 25 mg/l. Also, a very important point to make concerning this model (which we believe to be the most realistic one available) is that it is an engineered one, i.e., D, α, S_i, and X_R, the engineering parameters which can be controlled or for which engineering control can at least be approached have, in accordance with the model, been controlled. One cannot approach these results unless one approaches this degree of control. Also, it should be re-emphasized that the $\bar{S}$ refers to organic substrate in solution. Any kinetic biochemical model yet devised is addressed only to this aspect, i.e., the assumption is made that the biological solids have been separated from the mixed liquor before the effluent is discharged. If the separation is not essentially complete, the COD of the cells which remain in suspension will add to the values for effluent COD predicted by the model. Thus, although we can recommend the graphs of Figure 19 as a guide in selection of reactor detention time at rather steady loads of S_i, the model cannot replace the judgment inputs of the engineer. It can enable more enlightened and reliable judgment; it provides a way to balance rational kinetic theory and experimental data (lab and field experience).

Employing the model equations (43-47), one could construct his own "design" charts using biological constants for the specific waste being investigated. These can be determined as previously described (Chapter IV), using populations acclimated to the waste in question. Also, during the treatability study, the expected amount of biologically available carbon source (Δ COD) is determined. Then the behavior of $\bar{X}$ and $\bar{S}$ for various dilution rates and engineering control design parameters can be computed and charted. The resulting charts can be employed as the design guides. One of the most important things to be considered is that there needs to be some provision in the design to attempt, or approach, control of the controllable parameters, i.e., S_i, F, α, and X_R.

Possible ways and means of gaining some engineering control over variation in S_i and F will depend upon analysis of the individual waste water situation. For some industrially produced wastes, control of S_i and F may be possible at the various individual sources comprising the total waste. This is not always feasible, and in the case of municipal wastes, this type of control may be even more difficult to attain. Control at the treatment plant would seem to dictate use of some sort of surge basin. For a waste containing settleable material, it would be best to place the settling tank before the surge basin. There seems little to gain in using a surge basin as a settling basin. Depending upon the size of the surge basin required, the cost of providing for removal of sludge from the basin could be prohibitive. Also, depending upon the size of the basin, relative seeding population and strength of the waste, etc., some slight degree of mixing and aeration in the surge basin may be required. Thus, it seems best to let the settling tank ride on the flow in the usual manner and to even out F and S_i to the activated sludge tank. Microbial growth in the surge basin can be somewhat controlled by providing a paucity of aeration, e.g., enough to keep it from going anaerobic but not enough to allow the carbon source to limit growth. These are aspects which are naturally subject to engineering investigation in the treatability study stage which precedes the design stage. Some wastes will be grossly deficient in essential nutrients such as nitrogen and phosphorus, and this situation may obviate the need for providing aeration in the surge basin. If there is much growth in the surge tank, the biological solids concentration, X_i, entering the activated sludge reactor might be included as a term in the materials balance equation. This would lead to some modification in the resulting model equations (equations 39 through 47). In most cases, however, the amount of biological solids which grew in the surge tank would be small in relation to those in the aeration tank (which contains the recycle solids) and the assumption of negligible X in the aeration tank influent would appear to be satisfactory.

When the use of a surge basin is contemplated, it is advisable to determine μ_{max}, K_s, Y, and ΔCOD, using the effluent after the needed period of storage in the surge tank. Metabolic activity in the surge tank even under severe O_2 limitation can lead to chemical alteration of the carbon source in the waste. Thus, although significant substrate removal may not take place, the original organic carbon source may, in some degree, be converted to metabolic products (e.g., some fermentation products). This may have the effect of reducing the variability of the kinds of organic compounds in the waste. This slight amount of pre-treatment may be highly desirable (more research is needed here) but the point emphasized is that one should determine the characteristics of the system as they apply to the reaction vessel, and operations on the raw waste preceding the reactor can affect these characteristics; thus it is not necessarily the raw waste the designer is concerned with when considering the design of the activated sludge process.

Surge or equalization basins may or may not be considered in design depending upon the expected variation of S_i and waste flow rate in any particular situation. As we shall see in the next chapter, completely mixed systems are inherently provided with some degree of internal stability and do resist changes in $\bar{S}$ in response to changes in S_i and F. Control of X_R can be facilitated by the incorporation in the return sludge line of an aerated sludge consistency and storage tank from which sludge at constant concentration X_R is pumped at flow rate αF.

It might be argued that prolonged holding of sludge in the consistency tank might affect the biological constants. Some aerobic digestion could and would take place in this tank in which concentrated sludge is aerated. In this case, portions of sludge itself represent substrate, and preliminary studies indicated that even aerated storage up to 12 days did not cause larger variations in μ_{max} (when the sludge was used as seeding material in growth studies) than those normally obtained with freshly grown populations. A greater amount of experimental data on this aspect would be desirable, but it does not appear that the placement of an aerated sludge consistency tank in the return line will cause any drastic change in the biological constants.

Summary and Conclusions

Equations have been developed in this chapter for three kinetic models describing the steady state in continuous flow, completely mixed, microbial growth reactors. The first model considered was that for the simple once-through reactor generally used in microbiological research with pure cultures. Experimental evidence for the applicability of this model to "steady state" growth of heterogeneous populations was presented. Two models describing systems more operationally similar to the activated sludge process, i.e., continuous flow reactors with cell recycle, were then developed and compared with each other and with the once-through model.

Herbert's model, in which the ratio of recycle solids to aerator solids is maintained constant, was originally developed for use with pure cultures. Experimental testing of the model showed that it provided a fairly accurate prediction of the behavior of heterogeneous populations. However, operational control, even in the laboratory, was difficult and a steady state in solids concentration could not be achieved.

The kinetic model which we have proposed in equations 43 through 47 is more operationally feasible since the system constant employed is the concentration of recycle solids. While this model is not theoretically correct in that prediction is poor for conditions approaching dilute-out, it is realistic and is applicable over the range of values of the various kinetic parameters which might be encountered in practice in biological treatment. This model is recommended for use in the design of activated sludge processes, with the caution that provision for control of the parameters S_i, F, α, and λ_R must be made in the design and must be utilized in operation. The flow sheet for this model includes a sludge consistency tank required for maintaining constant recycle solids concentration, and a surge tank is recommended for systems expected to be subjected to significant variations in volume rate of flow or in waste concentration.

Both the surge basin and the sludge return consistency tank represent departures from "standard" or "traditional" practice. They represent changes which are in accord with more sound conceptual principles of microbial metabolism and accommodate the particular needs for engineering control of the process for the heterogeneous bio-mass, i.e., an activated sludge.

The kinetic model herein recommended and the suggested changes in unit processes which are desirably included in the flow sheet in order to best accommodate the model are of practical utility to engineers and operators. The model is by no means the "last word" concerning activated sludge, nor do we claim that it comprises the ultimate in kinetic theory. It is recommended simply as a practical advance in conceptual understanding and in design and operational control of the biological process.

VI

FACTORS TENDING TO DISRUPT THE STEADY STATE - SHOCK LOADINGS

In the previous chapter we presented a kinetic model for activated sludge of the completely mixed type in which the system is designed so that it can approach a steady state condition. The steadiness is encouraged by engineering actions in the design phase which are intended to smooth out fluctuations in S_i and F, and operationally the system is encouraged toward steadiness by controlling α and holding X_R, rather than c, constant. These are things engineers can do to make an activated sludge process operate more in keeping with the premise on which it was designed.

Even with the incorporation of these steadiness-enhancing modifications in the design of the system, there can be times when the activated sludge may be subjected to changing environmental conditions which tend to disrupt the steady state. Any environmental change tending to do this can be termed a shock load. Those environmental changes which cannot be, or have not been, smoothed by preventive engineering expedients must be accommodated solely by successful biological response or by combined biological and engineering remedial responses and/or measures. It is somewhat paradoxical that although the response of natural (heterogeneous) microbial populations to environmental changes has been one of the prime research interests of the senior author for more than a decade, the space which can be given to this subject in this particular document is the short chapter which follows. This chapter is intended to serve primarily as an introduction to biological responses to environmental changes, or shock loads. One can begin to appreciate the vastness and depth of this area by remembering that the biological world as we know it today evolved in response to environmental change. Often the environmental change was one which was imposed externally on the biological system and often the change in environmental conditions was produced by species within the "bio-mass." Adaptation and selection of species occur in response to environmental change. Also individual species can make an internal (molecular level) adjustment and become acclimated to a change in environmental conditions. An activated sludge, or for that matter any naturally selected microbial population, represents a microcosm which in itself can be said to serve as a crude model of the evolutionary process. Thus it possesses many of the complexities which, if ultimately resolved through continued investigation, could add in great measure to attainment of the ultimate goal of the life sciences, i.e., understanding of the biological world.

Whether or not one agrees with these statements there can be little argument concerning the mechanistic complexity of these responses, and oversimplification of the situation would be misleading and a disservice to the profession. On the other hand, in-depth discussion at the molecular and ecological level would require a voluminous document

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and would probably necessitate greater amounts of study and reference reading in the area of biology than most practicing engineers would be able to devote to it. While one could surely give valid arguments for such a course of study by pollution control engineers, it is not appropriate for the present document. We shall, however, attempt to introduce this important subject and set forth some rough and tentative guidelines which engineers can use. These are based primarily upon research in our laboratories, much of which is yet to be published.

Quantitative and Qualitative Changes in Substrate

A quantitative change in S_i (concentration of influent carbon source) is the type of perturbation usually and traditionally envisioned as a shock loading. In the usual jargon, a shock loading is envisioned as an increase in "BOD loading," although it could also be a decrease. A decrease in organic loading, if not too severe, e.g., a sudden halving of the loading, would precipitate some adjustment in X (concentration of biological solids) but would not be expected (in our experience) to cause a severe leakage in substrate (i.e., increase in S) in the effluent. Such shocks cannot be discounted, but our experience indicates that they are less deleterious than a comparable increase in loading.

An increase in loading can come on the line in various forms. There can be a rise in concentration of metabolizable carbon source with no change in F, i.e., a quantitative shock load. There can be a rise in organic substrate concentration which is accompanied by a change in F, i.e., a combined quantitative and hydraulic shock loading. Also there may be a rise in the total concentration of organic substrate coming into the system, and the additional organic matter may be of an entirely different type than the steady state substrate. Alternatively, the entire organic loading may change from one type to another, probably also involving a change in total organic concentration. In brief, the quantitative shock loading in S_i may often be accompanied by a qualitative shock loading in S_i. Also the change in S_i may be more or less gradual and of variable duration. Or it may be applied as a relatively short-lived pulse or slug dose. Slug doses to the sewer are commonly experienced and, where the run to the treatment plant is short, the activated sludge aeration tank may receive a relatively unattenuated slug. The use of completely mixed reactors is helpful in this regard because of the instantaneous dilution of the inflowing material (see Figure 10).

Thus it can be seen that even the "simple" quantitative shock loading (e.g., an increase in BOD loading) is not really a simple concept for which it would be easy to provide equations, "models," and/or rules of thumb of unquestionable predictive value. However, some approaches to conceptual analysis can be initiated, by considering the system which might be expected to be the most susceptible to deleterious changes in

S_i. Some conceptual insight into the ability of such systems to respond successfully to changes in S_i may form a basis for making conservative estimates for a more shock-resistant system. The least resistant system is the once-through system with no cell recycle. Activated sludge processes are not designed to operate as once-through systems, but the study of such systems provides conservative insight. The recycle of biological solids would be expected to provide resistance to the leakage of substrate in a shock-loaded process and, in our experience, it does. Therefore, systems without recycle are more suitable for studies designed to determine how each type of shock affects the steady state and limits within which the effects are acceptable.

A quantitative shock load occurs in a once-through system operating under steady state conditions of X and S (S_i and D constant) when the inflowing feed, S_i, is increased. If the increment of increase in S_i were not metabolized by the sludge, substrate concentration in the reactor (and the effluent) would increase along a dilute-in curve (Figure 10) until the new effluent substrate level would be equal to $\bar{S} + \Delta S_i$, i.e., the former level of $\bar{S}$ plus the new increment in substrate concentration due to the increase in S_i. The system would be said to have undergone a decided functional failure, i.e., the effluent substrate concentration would have increased to an unsatisfactory level. For a successful response to occur, S in the reactor would have to be held approximately at the steady state level of $\bar{S}$ by increased growth (increased X) due to consumption of the new amount of carbon source. Thus X would increase to a new steady state level, while S would remain the same or, in any event, the same percent substrate removal efficiency would be maintained. In accordance with the hyperbolic equation (Monod equation, 18) the increase in substrate can be expected to cause an increase in the specific growth rate, μ, thus increasing X. The increase in X uses up the new substrate (Y remains "constant"), preventing S from rising. The behavior of X and S, i.e., rise in X to maintain S constant, forms the transient stage of successful response as the system approaches the new steady state in accordance with the new concentration of S_i.

In an approximate way such a response can be shown to occur, but one must use "conceptual caution" tempered with empiricism and experimental observation in analyzing this important response. The hyperbolic (Monod) equation is in itself an approximation. It will be noted that in equation 18 we have employed S_o as it refers to batch (transient system) kinetics and have indicated that the substrate value, S, could be used provided it was understood that it referred to $\bar{S}$ in continuous flow systems under steady state conditions (equation 31). In a continuous flow system when S_i in the incoming flow is changed, steady state conditions no longer exist (because S is changed) and it has been shown that μ does not change instantaneously in response to a change in S (20, 21, 33). Therefore, equation 18 really cannot be expected to predict precisely the concentration of S and X during the transient resulting from an increase in S_i, although this relationship does provide rather good prediction of the new steady state levels of $\bar{S}$ and $\bar{X}$.

Figure 20 shows a response in X (upper portion) and in S (as COD and carbohydrate) when a once-through system, which was operating previously in a "steady state" with S_i of 450 mg/l COD, was shock-loaded by changing S_i to 1450 mg/l COD. The dilution rate, F/V, was 0.244 hr^{-1}, i.e., a reactor detention time of slightly over four hours. The response to this rather severe shock loading was an increase in X during the transient phase as X approached its new steady state level, but X did not rise rapidly enough to prevent substrate leakage (see bottom portion of the figure). Approximately 1.5 to two detention times were required before the system recovered, i.e., regained the steady state level of S. Obviously this shock was greater than the system could successfully accommodate, unless one accepts a six-to-eight hour functional disruption.

This severe shock loading was purposely applied so that COD would leak from the system in order that the Monod equation could be tested during the transient state. The computations involve numerical integration techniques applied to the balance equations in X and S (equations 25 and 28) for once-through systems (i.e., dX/dt and dS/dt not equal to 0 in the transient state). These calculations are discussed in reference 33. It is important here to note that the dotted curves for S and X labeled "predicted by the Monod equation" provide at best a rough prediction of the observed transient response.

An important consideration in response to shock loads is illustrated in the plot of filtrate carbohydrate COD (values designated on the figure by triangles). These values were obtained by analyzing filtered samples for carbohydrates (the synthetic waste being fed was a carbohydrate), and it is seen that during the transient rise in COD a very significant portion of the substrate which leaked from the system was not the original carbon source being fed but consisted of organic compounds produced by the organisms from the original carbon source. Thus, during the quantitative shock loading there was also a qualitative change in S. Even a "simple" quantitative shock can cause complications which can affect the mechanistic and kinetic response to the change in the environment.

It could be argued that attempts to provide mathematical models for depicting or predicting transient responses are, at this stage of the knowledge, somewhat futile because to cover all of the metabolic contingencies, they would need to be very complicated and unwieldly to employ. Any relatively simple model would have to be such a gross oversimplification of the system that it would not provide meaningful output. Obviously, there is need for much data obtained by study of defined and controllable environmental changes before satisfying mathematical models for shock loadings can be produced. Such endeavors are an important area for continued research. However, from the standpoint of the consulting engineer interested in the design of the process, mathematical models for the steady state operation are of more immediate utility, and what are vitally needed are guidelines regarding the extent of change in S_i which can be accommodated without severe disruptions of the

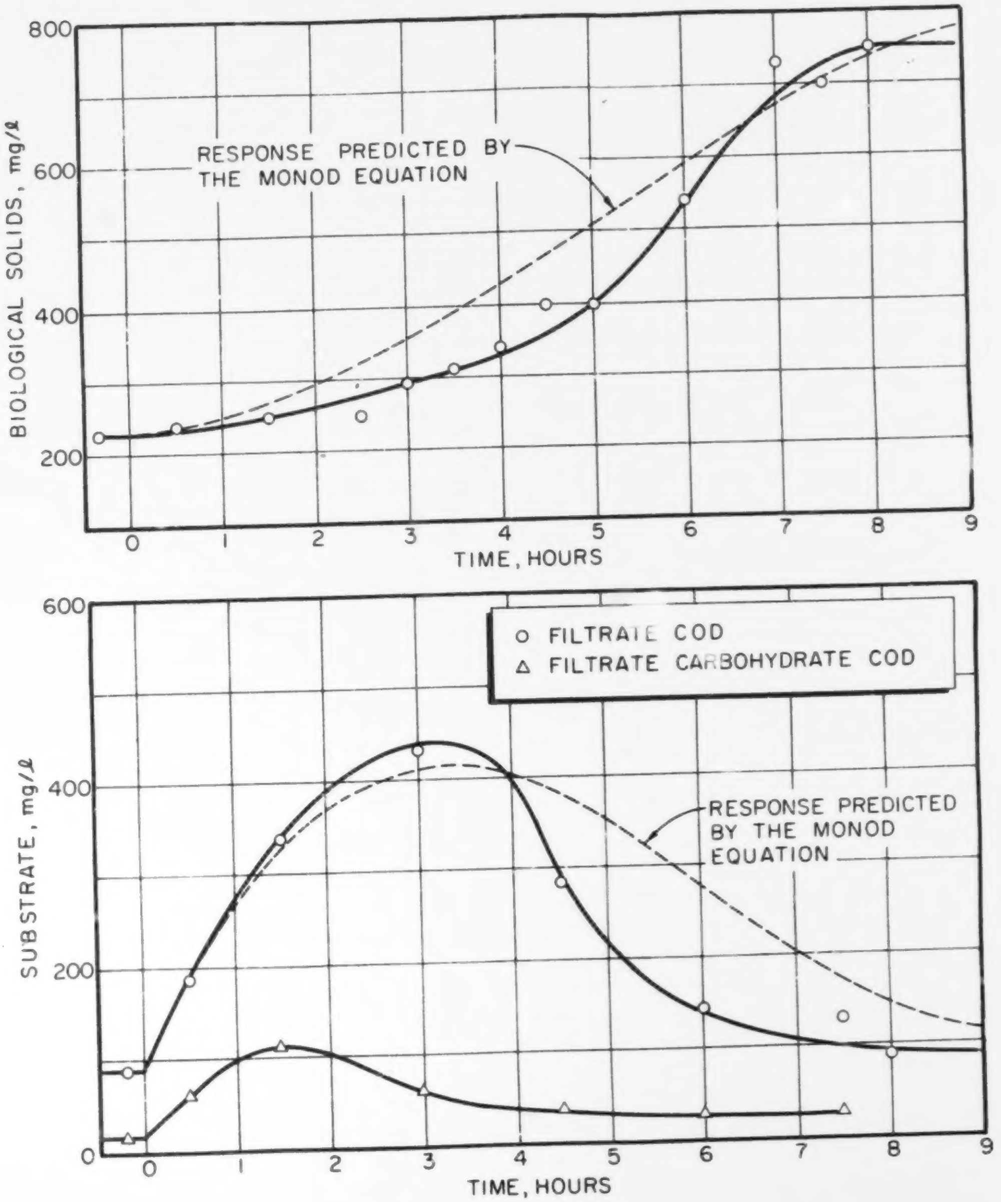

Figure 20. Response of a continuous flow, completely mixed reactor of the once-through type to a severe quantitative shock load consisting of a change in S_i from 450 mg/l to 1450 mg/l COD. The dilution rate was 0.244 hr^{-1} (33).

designed plant efficiency. Obviously, a 200% increase in S_i under operational conditions extant for the system of Figure 20 would not be desirable. The prime question which should be answered now is how much change in S_i could be accommodated without a rise in S. Even this information, while more easily provided than a mathematical model depicting the transient, is not easily given. We know from our own experiments that the change in S_i which can be accommodated is dependent upon the reactor dilution rate (or reactor detention time) and upon the biological solids level in the system. At reasonably selected reactor dilution rates, i.e., those not close to μ_{max}, even a once-through system will undergo little if any leakage of substrate during the transient state when S_i is increased 50% and the reactor detention time is approximately four hours. At a detention time of eight hours, a 100% increase in S_i can be expected to cause only a minor and short-lived increase in effluent S. With sludge recycling (maintaining X higher than in once-through systems), the range of increase in S_i which can be expected to be accommodated without a significant increase in S can be revised upward. We have obtained a large amount of experimental data for quantitative shock loadings at various detention times and recycle sludge levels and will be able, after more complete analysis of these data, to refine the ranges of increase in S_i which can be reasonably expected to be accommodated without system upset. For the present it seems reasonable, although somewhat conservative, to conclude that at detention times of six to eight hours one can allow increases in S_i of approximately 100% without expectation of significant loss of efficiency during the transient stage.

In Figure 20 the immediate metabolic response to a shock load in S_i was shown. This is the portion of the response which may eventually be depictable by some sort of kinetic model (for a recent attempt see the article by Young , et al., 34). There is, however, the possibility of ecologically important aftereffects which are manifested, not during the transient, but after the system has reached an apparent new steady state. These were uncovered during a recent investigation by Thabaraj and Gaudy (35), and Figure 21 is reproduced from that report. The data of particular interest in this figure are those in the lower graph showing biological solids concentration and substrate. This completely mixed continuous flow system (once-through) was operated at a dilution rate of 0.125 hr^{-1} (detention time of eight hours) with S_i = 1000 mg/l. It had attained a relatively steady condition and S_i was then changed to 2000 mg/l. A successful response to the change was registered, i.e., the biological solids "rose to the occasion" and S remained steady. The immediate (primary) response was one of successful accommodation to the 100% increase in S_i, and it appeared that a new steady state condition was being approached. However, between the thirtieth and fiftieth hours after changing S_i, there was a severe disruption in the steadiness of X and S, and the COD in the effluent (filtrate) rose to nearly 800 mg/l. The situation was not self-correcting until the seventieth hour after the shock (i.e., nearly nine reactor detention times). This secondary response was accompanied by a noticeable shift in predominating species

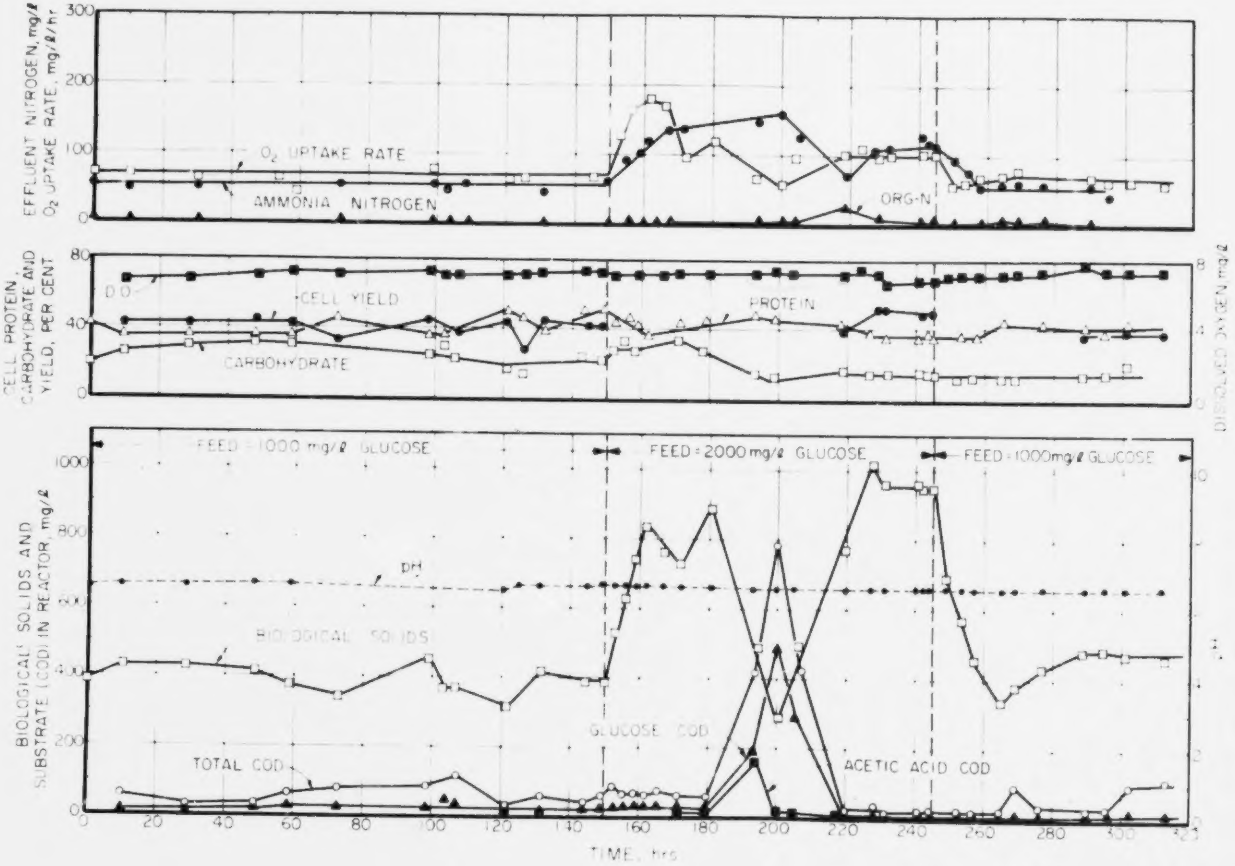

Figure 21. Response of a continuous flow, completely mixed system of the once-through type to a quantitative shock load consisting of a two-fold increase in S_i. The dilution rate was 0.125 hr^{-1} (35).

of microorganisms. The secondary response is one for which there may be little hope of developing any sort of reasonably reliable mathematical or "mechanistic" models, at least in the relatively near future. The possibility of this type of response does emphasize the need for caution and some conservatism with regard to engineering guidelines for allowable limits of increase in S_i.

In many cases the approximate allowable limit we have tentatively recommended, i.e., up to 100% increase in S_i, does not seem unduly difficult to attain by a combination of on-line procedures and installation of an equalization basin at the plant site.

Qualitative Shock Loads

Over the years, pollution control engineers have become accustomed to thinking of the incoming organic loading, i.e., BOD, as a specific entity. When it changed, as under shock loading conditions, one had more (or less) of the same kind of organic matter. Also, because of the diversity of the microbial population, as well as the diversity in the kinds and amounts of organic compounds in the wastes, one could aptly envision the ecosystem as one in which each species metabolized that portion of the incoming waste to which it was acclimated. Thus all component substrates in the waste were thought to be metabolized concurrently.

However, early studies employing mixed (heterogeneous) microbial populations in experimental systems in which two compounds comprised the organic substrate indicated that the presence of one substrate could prevent or hinder the metabolism of the other compound even though the population had previously been acclimated to it (36). In accord with the general beliefs previously outlined, this would not have been expected to occur with mixed populations.

It was known from basic research investigations with pure cultures that certain enzymes required to metabolize specific substrates are not present in the cell at all times (i.e., they are not constitutive enzymes) but are produced only when needed (i.e., they are inducible enzymes). The required inducible enzymes are thus produced by the cell in response to the presence of the particular organic substrate for which they are needed. The time required to turn on the genetic synthesizing machinery represents the acclimation period. If the organism is not genetically coded for this particular machinery, it cannot make the requisite enzymes and it cannot use the substrate. This is one of the ways in which changes in predominance of species are brought about in natural populations when changes in the available substrates occur. Whether an organism has the ability to acclimate biochemically determines its fitness for survival in the adaptation process. When one develops an "acclimated sludge," both selection of species (adaptation of the mixed population) and acclimation can occur. One of these processes, biochemical acclimation or synthesis of inducible enzymes, is an intracellular response, whereas the other, adaptation, is an inter-cellular or inter-species phenomenon. Both are important processes which are set in motion in response to qualitative shock loadings. In response to the qualitative change in S_i, the cells in the reactor can acclimate, or there may be an adaptation of the population or a combination of both. Both mechanisms of response may overlap or go on concurrently, with or without a disruption in system efficiency, i.e., leakage of organic substrates. On the other hand, these responses may occur in sequence.

Waste streams usually do not completely "turn over" during shocks; some of the former substrates are present with the new, and one would think that the former carbon stream would continue to be removed while the system attempted to accommodate to the new. This is, of course, one possibility, but experimental observations indicate that the reverse situation can also occur. That is, the new substrate may prevent (repress) continued synthesis of the enzymes required for metabolism of the former carbon source(s). The control mechanism by which the synthesis of one group of enzymes is blocked and another triggered has become known as metabolite repression, and a detailed discussion of this phenomenon is beyond the scope of the present report. However, at this point one may wonder concerning the functioning of the enzymes which have already been synthesized; cannot these continue to function? There is, after all, quite a large amount of acclimated sludge in the reactor when the change in S_i occurs. Should not these organisms be expected to continue functioning on the old substrates even though the new substrates may have prevented the continued production of enzymes? This was a question we posed as investigators and, in the process of experimentation, we came to the conclusion that there was, in addition to the blockage of the synthesis of enzyme systems, i.e., metabolite repression, another more immediately acting mechanism by which the functioning of the already existing enzyme system may be inhibited (either partially or completely) by the presence of new incoming substrate.

While the discovery of this type of control mechanism for enzyme systems in oxidative metabolic reactions contributed to the knowledge in the basic field, it also was of practical significance in the area of waste water pollution control, and a considerable experimental program was undertaken in our laboratory in efforts to elaborate both basic and applied insight into the phenomenon and the factors which might tend to attenuate or to exaggerate the magnitude of disruption ensuing because of changes in composition of the incoming waste. A considerable amount of these results have been published (3, 8, 36-46). However, much of the work remains to be published, and even more research is needed on this type of shock.

In view of the infinite variety of qualitative changes in inflowing substrate which can occur, it is impossible to predict the effects of such changes upon purification efficiency. It is, however, possible to provide some guidelines regarding factors which seem to attenuate disruption of system efficiency due to qualitative shock loadings. In some cases, the age or maturity of the sludge seems to provide an attenuating effect on the degree of interference which one substrate may exert on the removal of another or, in any event, on the manifestation of deleterious effects. Thus freshly grown, highly active cells, while admittedly exhibiting increased substrate removal capability, may be more susceptible to a change in the substrate composition than the older ecosystem which evolves upon longer retention of cells in the system. The employment of completely mixed reactors also helps to attenuate qualitative shocks since the change in incoming substrate is diluted into the system thus tending to permit more time for the system to generate a response (metabolic and ecological). The equalization basin, which was recommended for smoothing out variations in flow rate and concentration of substrate, will also be useful in preventing abrupt changes in the composition of the waste.

Changes in Chemical Composition of the Inflowing Waste Other Than the Organic Substrates

The incoming waste can undergo significant chemical changes involving, not the amount or kind of carbon source, but other chemical constituents which may affect metabolism of the carbon source. Among changes of this type, the effects of introducing toxic components, either inorganic or organic, are readily appreciated, and it is also readily appreciated that the best way of handling such situations is to require treatment of waste streams containing toxic components at the source of the waste water production prior to their entry to the waste stream influent to the biological treatment plant. Various wastes which would in general be toxic may be subjected to biological treatment if the toxicant stream is steady or controllable. For example, several decades ago phenolic wastes were considered toxic, but it is now well known that such compounds are amenable to biological treatment. An activated sludge can be developed for phenol metabolism, but the population "natural" to this environment can be expected to be a rather restricted one and would be predominant only if phenol was the predominant carbon source in the waste. If phenol were not the predominant steady carbon source, but were to come on and off the waste stream as a shock loading, it could inhibit many of the species developed in response to the dominant carbon source(s) and it would not be readily metabolized because the relatively fewer species which can metabolize phenol would not likely be present in the sludge when the shock occurred, i.e., the sludge would have to re-adapt to it each time. The same generality might be applied for considering inorganic toxicants, e.g. cyanides, chromium, etc. Sludges can be developed which exhibit increased tolerance to some of the inorganic toxicants, but again only when the system is fed a rather steady diet of the toxicant.

Certain normally nontoxic inorganic components, when applied to the system in high concentration as shock loads, can also cause severe disruption of substrate removal efficiency before the system can adjust. Sodium chloride is an example of compounds in this category. We have observed that activated sludges developed on relatively fresh water are considerably more resistant to shocks of high sodium chloride concentration than are sludges developed at high sodium chloride concentration to shocks with fresh water. Here again, it might be expected that the sludges developed in high salt concentration contain a narrower selection of species and, in general, a selection of species which may require salt (rather than, or in addition to, being tolerant to it). Thus, when the salt concentration is drastically reduced, e.g., to the fresh water level, these species die and undergo lysis, and during the interim transient period of adaptation, substrate leakage occurs. Since sodium chloride is not disruptive to biological metabolism at relatively low concentrations, only a rather drastic change would cause a deleterious transient response. For example, in a once-through, completely mixed system operating with an 8-hour aerator detention time, the efficiency of substrate removal was disrupted for nearly two days when the sodium chloride concentration in the reactor was increased to 30,000 mg/l over a period of one day (see Figure 1, reference 47). However, an increase to 8000 mg/l sodium chloride in the same period caused no disruption in substrate removal efficiency (see Figure 5, reference 47). It cannot be said that changes of the same magnitude, or even smaller changes, involving other inorganic constituents of the waste would not have more deleterious effects. In general, the effects of changes in the concentration and types of inorganic constituents of the carriage water should be the subject of continued investigation as the necessity for exertion of greater technological control of the aqueous environment increases.

Changes in pH also constitute chemical alterations of the growth environment which can cause severe transient responses. It is rather well known that systems can be adapted to pH levels considerably below or above neutral (i.e., pH 7.0) when the pH is held relatively steady. Here again it should be kept in mind that most biological forms function best at or near a neutral pH. When one develops an activated sludge at either a high (e.g., 8.5-9.5) or low (e.g., 4-5) pH, he restricts or narrows the diversity of species which can exist in the system and thereby may be narrowing the diversity or range of successful ecological response to all types of changes in environment, i.e., to all types of shock loadings, which that system can accommodate. If one can be satisfied that the particular waste stream in question will be subject to only small variations in chemical composition, pH, etc., operation of activated sludge processes at either high or low pH values may save some costs of neutralizing chemicals and devices for their addition, etc. However, pH control and the monitoring of pH in the process are among the more readily facilitated engineering controls which can be applied either manually or automatically, and the control of pH either at or near neutrality is certainly recommended from the standpoint of optimum function of the plant in steady state and in response to shock loading as well as from the standpoint of possible deleterious interaction between the waste and the materials of construction. A systematic study of response of completely mixed systems to changes in pH has been made in our laboratories and these results will be prepared for publication. The present aim is to provide guidelines for allowable variation in pH which should not produce serious disruption of system efficiency. In this regard, our studies indicate that for systems operated in the "steady state" at neutral pH (6.5-7.0) a change in pH, i.e., a shock load, consisting of either an increase or decrease of one pH unit can be accommodated with little or no disruption of the metabolic efficiency of the system. Systems can recover from more severe pH shocks but a transient leakage of substrate can be expected to occur as well as retardation of the flocculation and sedimentation of the biomass, especially in the case of acid shocks. In general, acid shocks can be expected to be more deleterious than alkaline shocks.

Changes in Environmental Conditions not Involving Chemical Composition of the Waste

Two major types of system shock loadings in this category are thermal shocks and hydraulic shocks. These, like the environmental changes discussed previously, engender biochemical and ecological responses within the activated sludge ecosystem which, if the shock is great enough, can lead to substrate leakage from the process. Concerning these types of shock, systematic studies of the transient behavior of completely mixed systems have been accomplished in our laboratory (more extensively with regard to hydraulic shocks), and these results will be readied for publication in the research literature. In this report we will attempt to provide some approximate guidelines for allowable limits of change which can be expected to cause minimal or no leakage of substrates. As with all types of shock loading, it is rather difficult to provide simple dependable rules of thumb because shock loadings may be imposed in combinations and what one observes concerning any specific type of shock may not hold when that type of shock comes on to a system in combination with another. Also, differences in the immediate past history of the system can engender different responses to the same shock. For example, the dilution rate (or detention time) at which the system is operating prior to a certain change in temperature may certainly affect the response. Also the steady temperature at which the system was operated prior to the change may affect the response. Furthermore, for a given dilution rate the rapidity with which the change in the reactor temperature occurs can affect the response. Thus the response to environmental changes is an area which cannot be overly simplified, and there are many facets to construction of a reasonably reliable set of conceptual guidelines for design and/or operation.

Regarding changes in temperature, we have examined the responses of completely mixed systems operated at detention times of four and eight hours at a steady state temperature of 25°C. Increases and decreases in temperature were applied. At an 8-hour detention time, the system could accommodate, with no excessive leakage of substrate, an increase of 10° (25 to 35°C) and with some, but not severe, substrate leakage, a decrease from 25 to 18°C. These were not immediate changes in temperature. On the downward side the drop in temperature was slightly in excess of 1^o per hour and on the upward side somewhat less than 1^o per hour. At the lower detention time, a change from 25 to 35°C could be accommodated without substrate leakage, but a change in temperature from 25 to 18°C caused a significant and long-lived leakage of substrate. Thus the bio-mass can respond somewhat more successfully to shocks of increased temperature than to those leading to temperature decreases. This seems reasonably predictable since increases in temperature (within reasonable biological limitation) increase the rate of metabolic reactions, thereby engendering more rapid response. Also, microorganisms capable of growth between 25 and 40° (mesophiles) are much more common than those capable of growth below 20°C (psychrophiles). Shocks consisting of increases to temperatures above 35°C could be assimilated by the system but not without severe transient leakage in substrate. As a rough tentative guideline, it would seem reasonable to recommend for a system operating normally at or around 25°C that the reaction liquor should be protected as much as possible from daily variations (i.e., changes in temperature taking place over a 24-hour period) of ± 10°C, with particular precautions being necessary to prevent sudden drastic decreases in temperature.

Since the reactor volume V is fixed, a change in F causes a change in D (or in detention time) and, as was seen previously, D is related to the exponential growth rate, μ. Thus a change in the rate of flow of incoming waste imposes a change in μ. How the system adjusts itself to this hydraulic change determines the response, i.e., the loss or maintenance of substrate removal efficiency during the transient period between initial and final steady states. There is a limit to the change in F from which the system can recover, since if D exceeds μ_{max} for a sufficient period of time, the solids will be completely diluted out. If a system which was formerly operating at a given (steady) flow rate with substrate concentration, S_i, is now shocked by an increase in F while S_i remains constant, it can be seen that not only has D been changed, but the overall average (daily) organic loading has also been increased simply because the rate of inflow of organic matter has been increased. The system has not undergone a quantitative shock loading (change in substrate concentration) but the daily organic loading has definitely increased. This is a rather severe type of "purely" hydraulic shock loading. Often hydraulic shocks are accompanied by a concomitant decrease in S_i, and at times the shock may be accompanied by an increase in S_i. We have examined responses of completely mixed systems to both general types of hydraulic shock under a variety of conditions (with various synthetic wastes employing specific carbon sources and mixtures of carbon sources, changes in dilution rate, etc.). In accordance with our purpose of providing tentative guidelines for accommodation of shocks on the basis of our experimentation, it is recommended that the process be protected from fluctuations in F (with no change in substrate concentration) larger than 100% for design reactor detention times of approximately eight hours. That is, if a system is being operated at a detention time of eight hours on the basis of the design average daily flow, protection (by way of an equalization basin or a bypass of flow to another unit) against a two-fold increase in F (leading to a decrease in detention time from eight to four hours) should be provided in the interest of ensuring reliable and steady delivery of the design substrate removal level. This precaution cannot be expected to provide absolute protection against disruption of the system but it can be expected that in this range of change in F (provided there is no change in S_i, temperature, etc.) the transient leakage of S will be rather short-lived and the system will rather rapidly recover its former steady state level of S. Again, as a rough guide for design detention times less than eight hours, changes in F somewhat less than 100% should be protected against, and for design detention times greater than eight hours, changes in F somewhat higher than 100% may be allowable.

Summary

This chapter has served as an introduction to various types of shock loadings involving chemical changes, both quantitative and qualitative, of the incoming carbon source, changes in the chemical composition of waste water constituents other than the carbon source and physical changes such as temperature and volumetric rate of flow. The behavioral response of the bio-mass (activated sludge) in completely mixed continuous culture reactors is a complex subject, and in-depth understanding of transient kinetics may rightfully be a subject of continuing basic environmental engineering research. Before any sort of useful mathematical models for the transient response will be forthcoming, the molecular and ecological physiology of these systems will need to be penetrated. These problems are of enormous future interest to the entire area of engineering management of the aqueous environment. However, the most immediate concern of the design engineer is not the ways and means to predict transient responses, but ways and means to prevent deleterious transient responses leading to substrate leakage and guides to the extent of environmental change (shock loading) the process can accommodate without the need for preventive procedures. Thus we have attempted to provide rough but quantitative (and conservative) guideline limits for changes which the system can be expected to accommodate without serious metabolic upset. From our experimental observations, it appears that at reasonable design aerator detention times (six to eight hours), periodic increases in S_i of 100% of the average daily design concentration can be accommodated without serious or long-lived leakage of substrate in the effluent. Concerning changes in F, one can expect that hydraulic overloads representing a change of 100% in the average daily design flow (with no change of organic carbon concentration, S_i) can be accommodated at detention times of approximately eight hours. Where variations larger than these are expected or cannot be controlled in-line or at the waste source, the use of an equalization tank ahead of the aeration tank seems advisable to smooth out peaks above this range. Equalization tanks or basins for high volume plants (e.g., those in large metropolitan areas treating combined municipal and industrial discharges) may not be practical, depending upon the variation in flow. However, the range of change in flow rate for larger systems is considerably less than for smaller ones.

Effective control of recycle solids concentration, X_R, can also provide a useful tool for attenuating effects of shock loadings. One relatively reliable and easily understood concept is that, in general, the higher the concentration of cells, the more organic matter can be removed in a fixed time. Activated sludge plants are usually operated with much higher concentrations of biological solids than of substrate. For example, an organic concentration in the waste of 200 mg/l "BOD" and a concentration of biological solids in the aeration tank of approximately 2000 mg/l are not uncommon. Even allowing for the fact that we cannot say that all of the biological solids represent substrate users, a substrate to cell ratio of this magnitude indicates that activated sludges are essentially run under starvation conditions. This situation represents a potential reserve capability which can be drawn on during increases in S_i. The reserve capacity that is found in excess metabolic capability might also be expected to be beneficial during increases in F which decrease the reactor detention time. However, it must be remembered that the hydraulic shock loading which is applied to the aeration tank is also passed along to the secondary clarifier, and the decreased settling time may be more of a problem than the decreased aeration time. It may not only cause loss of solids in the effluent, but may also decrease the solids concentration in the sludge recycle flow. Thus the combined use of equalization basins and facilities to control X_R (see Figure 16) can help to steady the system under both normal and shock loading conditions.

VII

PROCESS MODIFICATIONS FOR NITROGEN-DEFICIENT WASTES

Nitrogen Supplementation for Biological Treatment

In previous chapters, removal of organic material from waste waters and growth of microorganisms have been treated as interdependent processes. It was stated that it is the usual practice in biological treatment to add nutrients such as nitrogen and phosphorus to wastes deficient in these elements to ensure that the carbon source is the limiting nutrient. This procedure has been recommended for maximum purification efficiency and it is a logical procedure since removal of the oxygen-demanding organic components of the waste has traditionally been, and still is, the foremost goal of biological treatment. The recommended levels of supplementation are based upon the ratios of carbon, nitrogen, and phosphorus required by microorganisms for "balanced" growth. This means simply that if one considers synthesis of microbial cells, as described in Chapter III, as the essential mechanism by which organic material is removed from the waste, the amounts of nitrogen and phosphorus required are dependent upon the amount of organic carbon which is to be removed. Based upon studies of optimum nutritional conditions for purification, ratios of 20:1 for BOD:N and 100:1 for BOD:P are commonly accepted and supplementation is provided to achieve these ratios if necessary.

The addition of nutrients represents a continuous operational cost which may amount to a significant fraction of the total cost of operation. Any means of reducing the supplementation required could therefore allow a significant reduction in the operational cost. An additional reason for re-examination of the practice of adding excess nutrients is the recent concern over stream enrichment, i.e., the concern that leakage of nitrogen and phosphorus from treatment plants may contribute to excessive algal growth in receiving waters. While much of the nitrogen and phosphorus in many waters is undoubtedly contributed by natural and agricultural runoff and while no nitrogen is required for growth of blue-green algae which are capable of using atmospheric nitrogen, it is important that all possible sources of nutrient addition be curtailed. For wastes which contain nitrogen in excess of the required BOD:N ratio, e.g., municipal wastes, the problem is removal of the excess nitrogen. However, wastes which are initially deficient in, or devoid of, nitrogen, e.g., many industrial wastes, present a different problem. For these wastes it is necessary to avoid adding excess nitrogen which may leak from the system while maintaining maximum purification efficiency with regard to removal of organic material. In a laboratory pilot plant or an industrial fermentation process using a pure culture and medium of known and constant composition, it would theoretically be possible to exactly balance the amounts of carbon and nitrogen sources so that each would be maximally consumed. However, on a practical basis using whole wastes which vary in composition both qualitatively and quantitatively,

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this is an impossible achievement, and it is necessary to consider other means of avoiding leakage of added nutrients.

Studies of the applicability of the Monod equation (equation 18) to the relationship between nitrogen concentration and growth rate of heterogeneous populations have shown that the relationship provides a good approximation of the data (48). Values for maximum growth rate, μ_{max}, were comparable to those measured in studies of carbon source limitation (see Chapter IV). The values of K_s ranged from 1.5 to 4.0 mg/l. Based on comparison of amounts of carbon source and nitrogen source required for maximum specific growth rate determined in these and other studies, a ratio of BOD:N of 25:1 appeared to be optimum for maximum growth rate. This indicates a nitrogen requirement only slightly less than that found in earlier studies using different methods (49).

The studies cited above were performed in batch systems. Rather extensive studies of the effect of nitrogen level upon the performance of once-through, continuous flow, completely mixed systems have also been made in our laboratory (50). The laboratory pilot plant was operated at various dilution rates with a synthetic waste containing 1060 mg/l glucose COD, and three levels of nitrogen supplementation were employed. These studies showed that purification efficiency and leakage of nitrogen in the effluent depended not only upon the level of nitrogen supplementation, but also upon the dilution rate. The nitrogen content of the sludge was also determined in these studies since this parameter appears to be related to the purification capability of the sludge under some conditions (49, 51).

Figure 22 shows the variation in effluent COD at three levels of nitrogen supplementation and various dilution rates, and Figure 23 shows ammonia-nitrogen levels in the effluent under the same conditions. It is apparent from Figure 22 that at longer detention times there is much less effect of the COD:N ratio on purification efficiency than there is at shorter detention times. At a detention time of twelve hours, fairly good purification was achieved, even in a once-through system, at a COD:N ratio as high as 70:1. (In a system with solids recycle, one might expect greater purification efficiency under otherwise equal conditions.) Very little additional purification efficiency was gained by decreasing the COD:N ratio from 40:1 to 25:1. With increasing detention time, nitrogen leakage increased sharply at a COD:N ratio of 25:1, while no leakage was observed at any of the dilution rates at a COD:N ratio of 70:1.

Figure 24 shows the relationship between nitrogen content of the sludge and purification efficiency at four COD:N ratios and various dilution rates. As these data show, even with severe nitrogen limitation, i.e., at a COD:N ratio of 70:1, the nitrogen content of the sludge can vary depending upon dilution rate. It is also apparent from this figure that the same purification efficiency may be achieved with sludges of

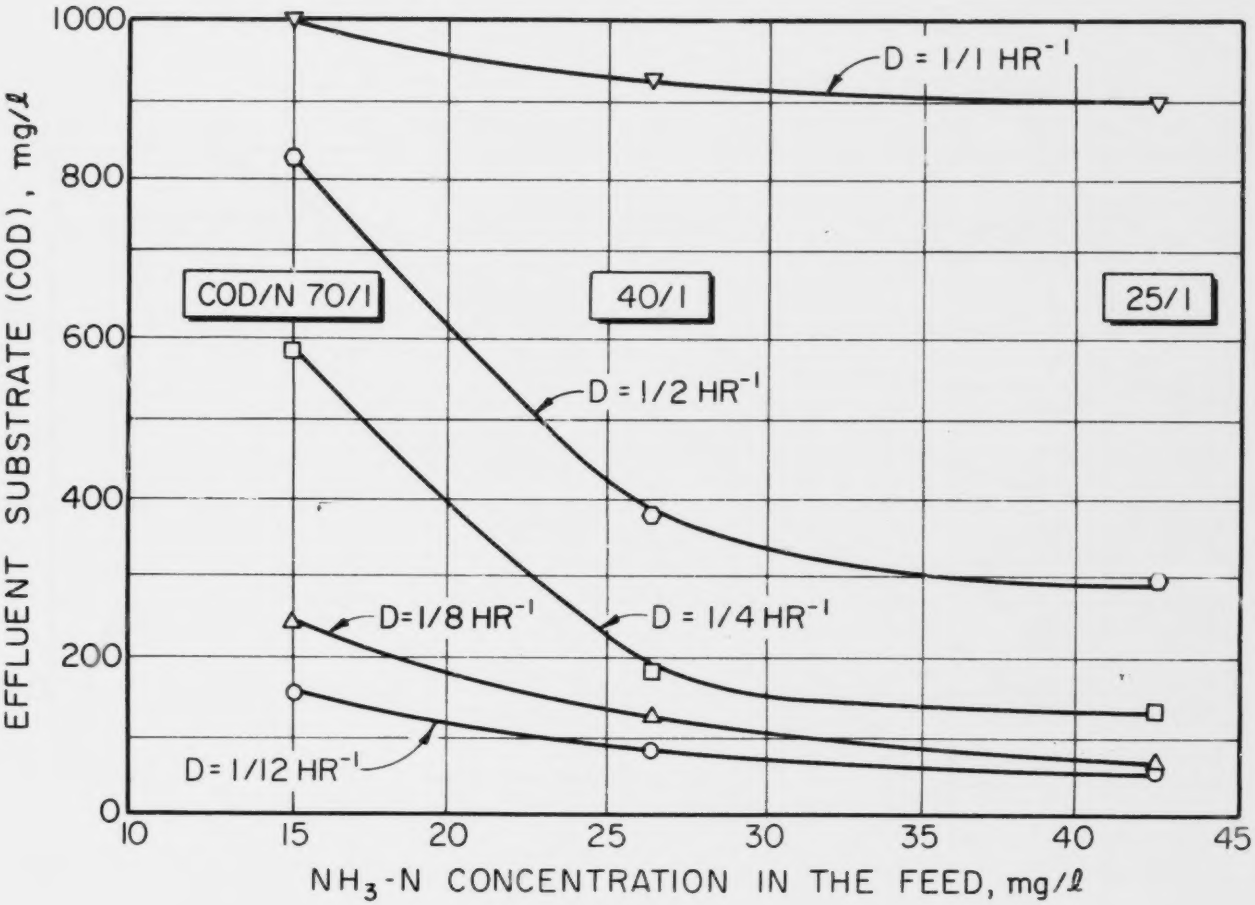

Figure 22. Relation between effluent substrate concentration, $\bar{S}$, and NH$_3$ - N concentration in the feed at three COD:N ratios for dilution rates ranging from 1 to 1/12 hr^{-1}. Feed COD was 1060 mg/l (50).

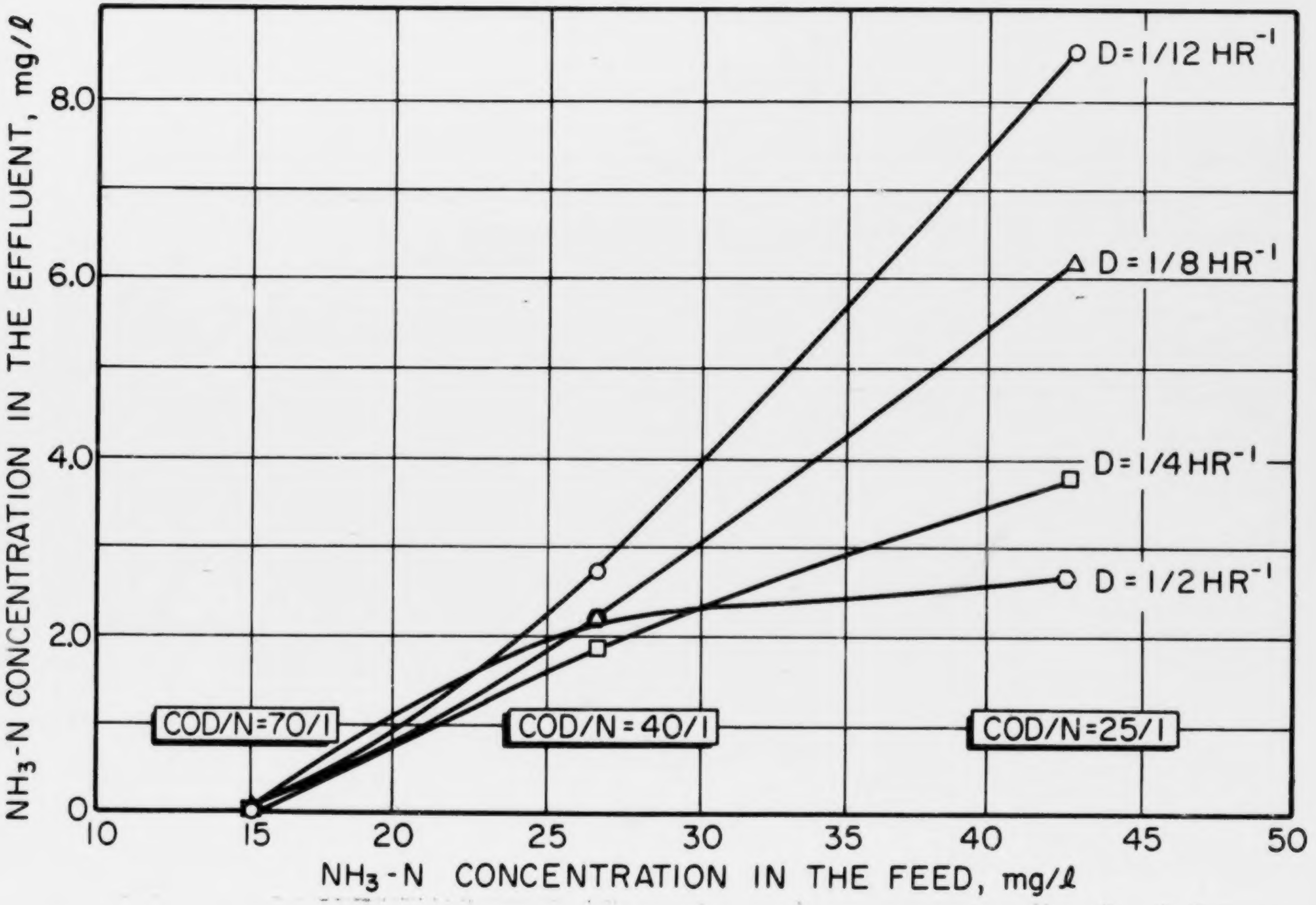

Figure 23. Relation between effluent and influent concentrations of NH$_3$ - N at three COD:N ratios for dilution rates ranging from 1 to 1/12 hr^{-1} (50).

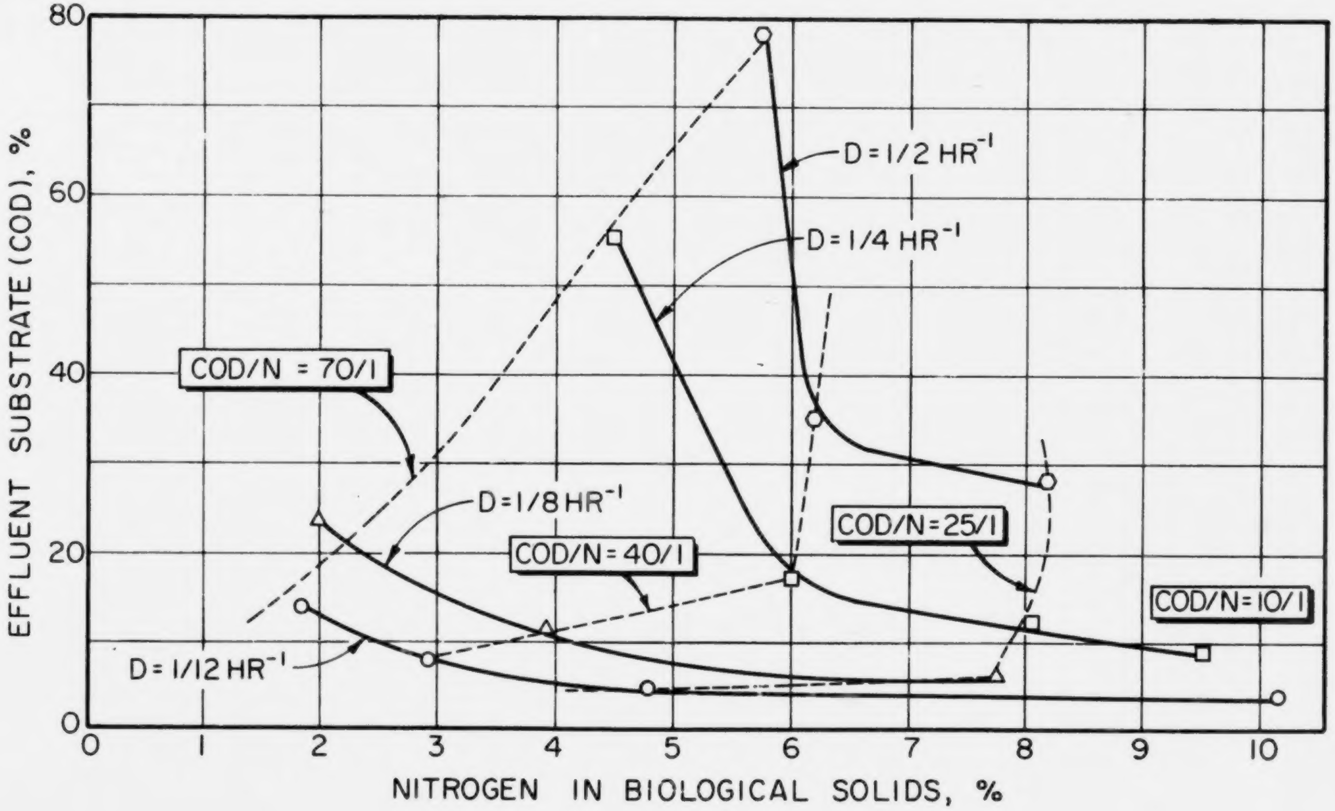

Figure 24. Relation between purification efficiency, nitrogen content of the sludge, dilution rate, and feed COD:N ratio (50).

quite different nitrogen content when other design parameters are varied. For example, 90% COD removal might be chosen as the desired purification. From the figure, one can determine that this degree of purification could be achieved with: a 12-hour detention time at a COD:N ratio of between 70:1 and 40:1 and a sludge nitrogen content of 2.5%; an 8-hour detention time, COD:N ratio of 40:1, and sludge nitrogen content of slightly above 4%; or a 4-hour detention time, COD:N ratio between 25:1 and 10:1 and sludge nitrogen content of 9%. However, concern over stream enrichment dictates that consideration also be given to efficient utilization of nitrogen, and the data shown in Figure 23 indicate that the lower COD:N ratios which enhance sludge nitrogen content also enhance nitrogen leakage in the effluent. Therefore, it should be emphasized that no single factor such as a standard ratio of nitrogen to BOD or COD or sludge nitrogen content can be chosen as the basis for design. Both COD and nitrogen utilization must be considered in choosing the design detention time and the level of nitrogen supplementation to be employed. It should also be pointed out that a system in which solids are recycled would be expected to require less nitrogen than a once-through system where cells are growing more actively and that nitrogen leakage would probably be greater in a system with recycle. Treatability studies carried out in the design stage should include determination of effluent nitrogen levels. Especially for industrial wastes which are not devoid of nitrogen but contain available nitrogen at a level below that usually considered optimum, i.e., wastes for which the BOD:N ratio is higher than 20:1, treatability studies should be designed to minimize or possibly eliminate nitrogen supplementation. For wastes which contain little or no nitrogen, a completely different mode of treatment may be preferable.

Our studies of nitrogen requirements for the activated sludge process suggested the possibility that substrate removal might be accomplished by oxidative assimilation without the addition of nitrogen. Concurrent studies on the kinetics of purification in relation to initial biological solids level (52, 53) showed that the COD: solids ratio is important in determining the mechanism of substrate removal. These findings led us to suggest a possible means of utilizing oxidative assimilation to effect a savings in nitrogen supplementation and avoid the possibility of nitrogen leakage while maintaining purification efficiency (54). The new modification of the activated sludge process which we have proposed for nitrogen-deficient wastes is described below. This rather radical departure from traditional methods of treatment of such wastes has been described in a series of publications (54-60) and some of the data obtained in laboratory pilot plant studies will be presented herein. Before discussing the process itself, it is desirable to describe oxidative assimilation and to discuss some of the basic kinetic studies which led to the proposal of the new process.

Oxidative Assimilation

In a system containing metabolizable organic compounds and other required inorganic nutrients, but devoid of an external supply of nitrogen, replication of microbial cells cannot occur because synthesis of essential proteins, nucleic acids, and some lipids requires nitrogen. However, many bacteria are capable of converting a large fraction of the available organic material into internal storage products, while oxidizing the remainder of the organic matter to provide energy for the synthesis of the storage product and for other activities of the cell. The specific product synthesized varies with the bacterial species. The most common internal storage product is a polymer of glucose which is very similar to glycogen, the storage product synthesized by the human liver. Other organisms synthesize a lipid polymer, poly-β-hydroxybutyric acid. These storage products may accumulate to a level of 50% of the dry weight of the cells. Some organisms, if inorganic phosphate is present in excess, also form a polyphosphate which serves as a store of both phosphate and energy rather than as a carbon source store. The storage of polyphosphate is undoubtedly responsible for much of the biological phosphate removal which can occur on prolonged aeration of activated sludge if the organisms which possess this ability are present in sufficient numbers. Other organisms may synthesize external polysaccharide which is deposited as a capsule or slime layer outside the cell wall. Upon addition of nitrogen to the system, internal storage products are utilized for growth.

Gaudy and Engelbrecht (61) studied removal of COD by activated sludge in the presence and absence of an external nitrogen source and showed that oxidative assimilation could occur in such systems, i.e., COD was removed in the absence of a nitrogen source, and the organic material was stored primarily as carbohydrate. These studies were carried out in batch at a solids:COD ratio of approximately one, using sludge harvested from the effluent of a continuous flow system. Interestingly, analyses of protein and carbohydrate content of the sludge throughout the period of aeration in batch showed that both systems, with and without nitrogen added, initially synthesized carbohydrate. In the system containing nitrogen, the carbohydrate content of the sludge later decreased while the protein content increased.

In later studies of the effect of initial biological solids concentration upon the kinetics of COD removal and solids synthesis, it was found that as initial solids level was increased in batch systems, the autocatalytic curves for both COD and solids typical of low initial solids systems (see Figure 1) became more nearly linear. That is, reactions which exhibited first-order increasing kinetics at low initial solids concentrations approached zero-order kinetics or, in some cases, first-order decreasing kinetics, when high initial solids were used (52, 53). Since replication of microbial cells is an autocatalytic process, simply because replication occurs by division of one cell to form two cells, the kinetics observed in these studies were atypical for replicating systems. This suggested that replication (cell division) actually does not occur to an appreciable extent during the period of substrate removal in a batch system when the ratio of initial solids to COD is high. Krishnan and Gaudy (53) showed that, at an initial COD:biological solids ratio of approximately 0.5, COD was efficiently removed in the presence of an inhibitor which completely prevents replication of cells by preventing synthesis of proteins. Figure 25 (taken from this publication) shows the course of COD removal and synthesis of biological solids, both of which approximate zero-order kinetics during most of the substrate removal period. These data demonstrate that replication of cells is not necessary for COD removal. Sludge synthesis in this case does not represent synthesis of new cells but rather an increase in the weight of the cells due to synthesis of storage products, primarily carbohydrate as shown by the analysis of the sludge. More detailed studies of rates of COD removal and synthetic reactions during the period of substrate removal in relation to the initial COD:solids ratio, provided more complete evidence that as the initial COD:solids ratio is decreased, COD removal and sludge synthesis follow more similar patterns during the period of COD removal in systems with and without nitrogen (62). In short, the mechanism of COD removal in batch systems containing high initial levels of biological solids is primarily oxidative assimilation, not cell growth. Subsequently, if nitrogen is present, the stored carbohydrate disappears and protein is synthesized, i.e., the storage products are converted to new cellular components required for replication (see Figures 3 and 4 of reference 53, and Figure 3 of reference 62).

This behavior of high solids batch systems may be explained as follows: When microbial cells are placed in fresh medium, replication is not initiated immediately unless the cells are from a population actively growing in the same medium. In most cases, e.g., when a culture is grown to maximum numbers and a portion is transferred to fresh medium, a period of adjustment is required before replication can begin. During this period, the lag period, the cell synthesizes the proteins and other cellular components required for growth under conditions which differ from those in which the cells had previously existed. The longer the period between cessation of growth and transfer to fresh medium, the greater is the amount of adjustment required, in part because enzymes no longer used once the food supply has been exhausted may be degraded by endogenous metabolic reactions. The cells used in the batch experiments described above were taken from batch activated sludge systems maintained by feeding once in twenty-four hours with one-third of the sludge and two-thirds of the supernatant wasted each day before feeding (52, 53). The wasted sludge was used for experimentation. This sludge was certainly not composed of young, actively growing cells. Thus, when the sludge was contacted with fresh waste, it was not capable of immediate replication. However, it was capable of removing COD, as our studies showed, and indeed it is necessary that COD be removed to supply energy for the formation of cellular components required for replication. As our studies have also shown, the sludge is, in addition, capable of forming storage products which it may later use as a carbon

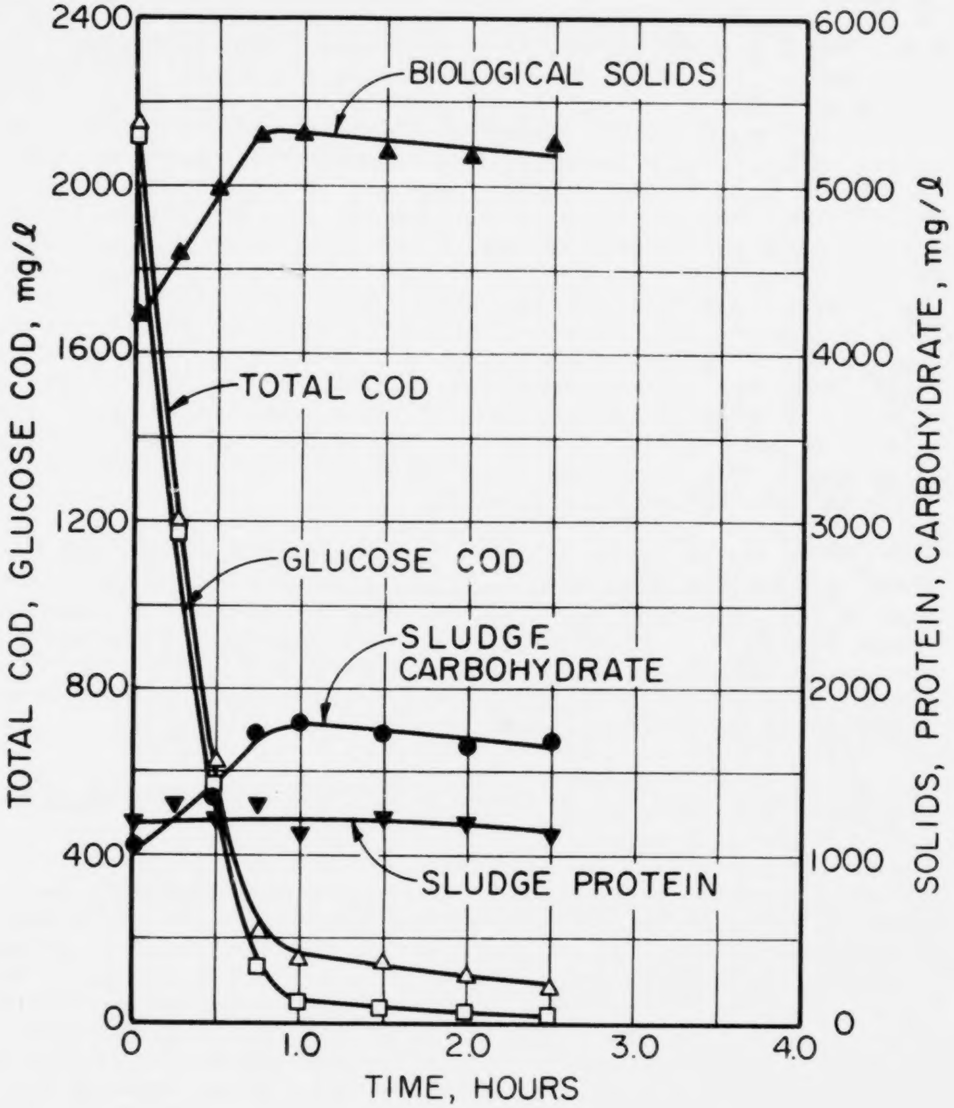

Figure 25. Substrate removal, sludge accumulation, and sludge composition in the absence of protein synthesis in a batch reactor (53).

source for synthetic reactions. Thus, sludge synthesis (increase in weight of biological solids, X) is not impeded but replication of cells is delayed. If only small numbers of cells are present (low initial biological solids), a significant amount of COD cannot be removed during the lag phase and COD removal is dependent upon formation of new cells, giving rise to the autocatalytic curves for substrate removal typical of a replicating system. However, if a sufficient number of cells is present in relation to the available carbon source, all of the carbon source can be removed even before replication begins. If large numbers of cells are present, the carbon source can be exhausted during the oxidative assimilation phase which precedes replication, and therefore replication may not occur, and is not required, for COD removal at low COD:solids ratios. However, for this process to be repeated, it is necessary that a nitrogen source be available so that the storage products may be utilized for synthesis of normal cell components. In batch systems containing very high solids, this conversion can take place after the purification phase is complete, i.e., after the carbon source (COD) has been taken up by the cells.

Since the purification of the waste, i.e., removal of COD, and the synthesis of nitrogen-containing cellular components can occur sequentially even in the presence of nitrogen, it should be possible to separate these phases physically. It seemed that several advantages might be realized by such a treatment scheme for nitrogen-deficient wastes. Accordingly, studies were undertaken to determine the feasibility of the process described in the following section.

Proposed New Process for Treatment of Nitrogen-deficient Wastes

The flow sheet for the proposed method of treatment is shown in Figure 26. After primary settling or other required pre-treatment, the nitrogen-deficient waste flows into the feeding aerator, or activated sludge tank, where it is contacted with a high concentration of recycled sludge in a completely mixed reactor. In contrast to the usual mode of operation, no nitrogen source is added, although phosphorus should be added if necessary. The waste is purified by oxidative assimilation; that is, the organic material is removed from the waste and converted to intra-cellular storage products. The mixed liquor exiting the aerator is settled in the usual manner (clarifier) and the supernatant is discharged as plant effluent. At this point, the settled sludge differs from the usual activated sludge in that it contains a higher percentage of carbohydrate and possibly lipid, and a lower percentage of protein. The effluent should be free of nitrogen, since the nitrogen-starved cells would remove any trace of nitrogen present in the waste. The waste has been purified without nitrogen having been added.

Before the sludge can be recycled to the aerator, its ability to convert external organic carbon to internal storage products must be regenerated by allowing the cells to utilize the material already stored for

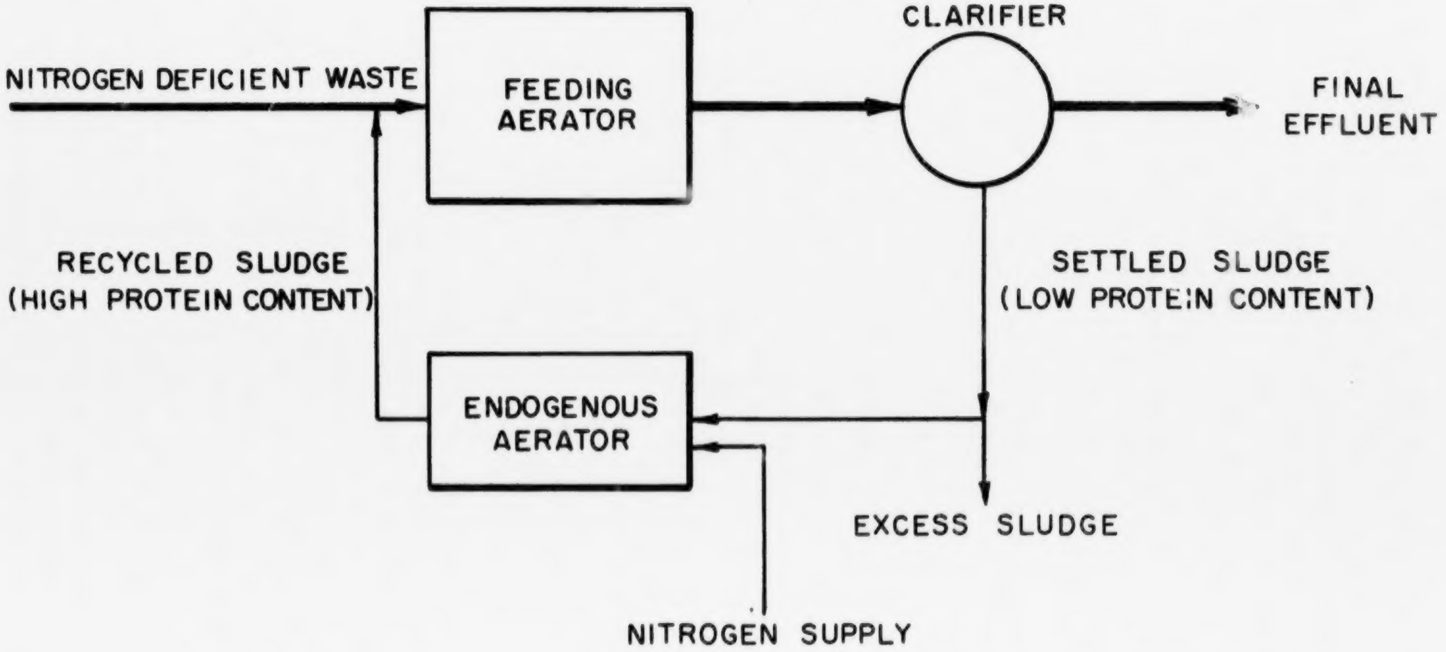
NITROGEN DEFICIENT WASTE
FEEDING AERATOR
CLARIFIER
FINAL EFFLUENT
RECYCLED SLUDGE (HIGH PROTEIN CONTENT)
SETTLED SLUDGE (LOW PROTEIN CONTENT)
ENDOGENOUS AERATOR
EXCESS SLUDGE
NITROGEN SUPPLY

Figure 26. Proposed flow sheet for the treatment of nitrogen-deficient industrial wastes by the continuous oxidative assimilation modification of the activated sludge process (54).

synthesis of the nitrogen-containing compounds (proteins, nucleic acids, and some lipids) which are required for continued metabolic activity and for replication. However, only that portion of the sludge which is to be recycled requires regeneration, and the amount of nitrogen required is therefore less than that usually added to allow unrestricted synthesis of nitrogenous compounds by all of the sludge in the aerator. The portion of sludge to be recycled is pumped to a second aerator, called the endogenous aerator because no external carbon source is present. Nitrogen is added, as NH_4^+, and the stored carbon source is used by the cells for synthesis of nitrogenous compounds, largely protein, and for replication. This regenerated sludge, now depleted of storage products and high in protein content, is recycled to the feeding aerator, where it again removes COD by oxidative assimilation.

Initial tests of the feasibility of this mode of treatment were carried out in batch systems using a synthetic waste containing glucose as the sole organic carbon source (54). Parallel systems were operated in an identical manner except that nitrogen (as NH_4^+) was added to the feeding aerator in one system and to the endogenous aerator in the other. The necessity for regeneration was also tested by recycling sludge from two endogenous aerators, to only one of which nitrogen was added. All sludges were analyzed during each aeration period (feeding or endogenous) for carbohydrate and protein content. These studies showed that the physical separation of waste purification and replication of cells was possible in practice. COD was removed with approximately equal efficiency in the presence and absence of nitrogen in the feeding aerators, and was stored primarily as carbohydrate by the sludge in the aerator to which nitrogen had not been added. In the endogenous aeration period, the stored carbohydrate was converted to protein in the presence of ammonia-nitrogen, and the recycled regenerated sludge removed COD as efficiently as the recycled sludge which had been treated in the usual manner by addition of nitrogen to the waste before treatment. Without the regeneration step, the sludge rapidly lost its ability to remove COD.

Since glucose is rather easily polymerized to form the glycogen-like carbohydrate storage product, it was important to test the feasibility of the process for other carbon sources. Batch studies (56) were carried out using synthetic wastes containing a non-carbohydrate, acetic acid, and lactose, a compound which is metabolized by inducible enzymes. The use of lactose was designed to determine whether inducible enzymes necessary for oxidative assimilation would be degraded during endogenous aeration in the absence of the carbon source. The process operated successfully with both synthetic wastes.

The feasibility of the process for continuous flow operation was studied in a laboratory-scale pilot plant (Figure 27) using two synthetic wastes and a whole waste. In the synthetic wastes, glucose (57) and acetic acid (58) were used as carbon sources, and the whole waste was a sugar refinery plant effluent obtained from the Imperial Sugar Company refinery at Sugarland, Texas (55). The nitrogen supplement for the endogenous

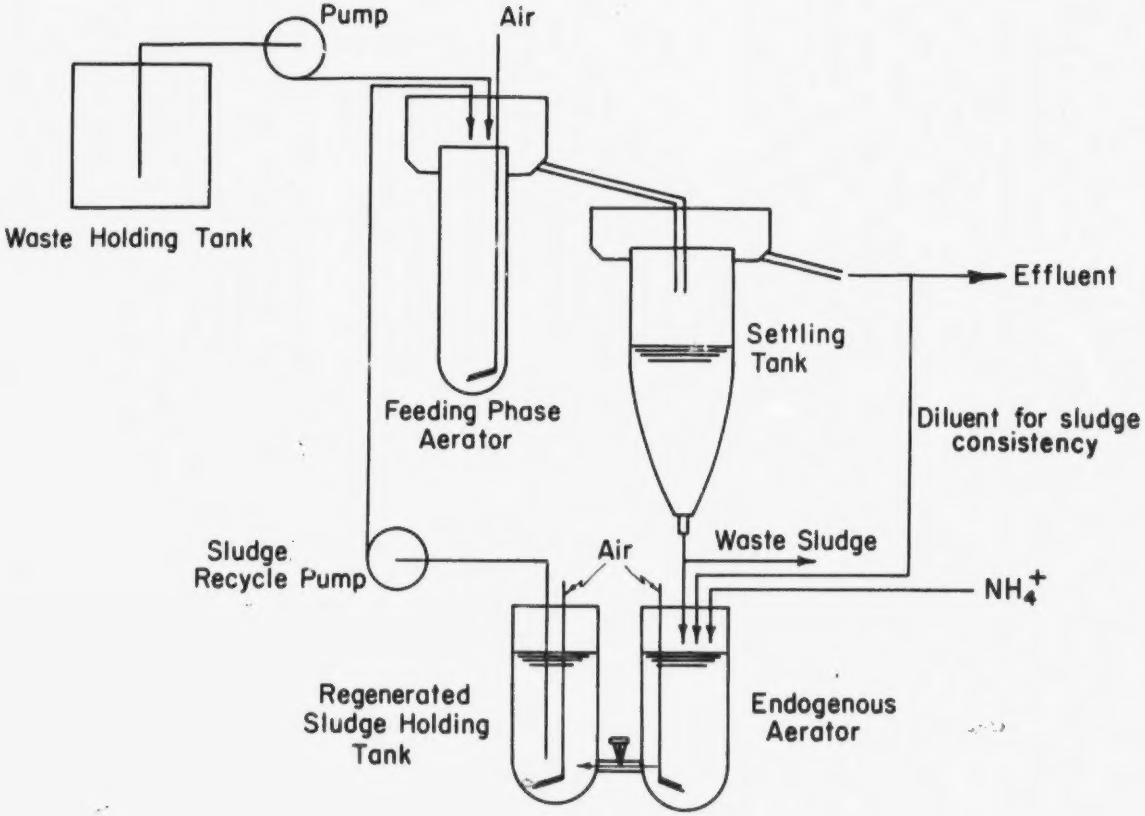
Pump
Air
Waste Holding Tank
Effluent
Settling Tank
Feeding Phase Aerator
Diluent for sludge consistency
Sludge Recycle Pump
Air
Waste Sludge
NH_4^+
Regenerated Sludge Holding Tank
Endogenous Aerator

Figure 27. Laboratory-scale pilot plant employed in operational feasibility studies for the continuous oxidative assimilation process (55).

aerator was calculated as a ratio of the COD added to the feeding aerator, and various ratios were used to determine the minimum requirement for the process. These data have been published in detail in the references cited above. With nitrogen-free glucose synthetic waste (1060 mg/l COD), 95% removal of COD was accomplished with a 4-hour detention time in the feeding aerator at a COD:N ratio of 70:1 (approximately equivalent to BOD:N = 50:1). Sludge wasted was 17%, and no NH_4^+ could be detected in the effluent. With acetic acid (1060 mg/l COD) at the same COD:N ratio and feeding aeration time, COD removal was 90%. Under the same conditions, 94% of the COD was removed when the COD:N ratio was 50:1. The sugar refinery waste varied considerably in COD content, and no attempt was made to adjust its concentration since this variation allowed assessment of the ability of the process to withstand quantitative shock loads. The COD of the waste varied from 380 to 1375 mg/l, and BOD from 271 to 841 mg/l. Approximately 35% of the BOD of the waste was carbohydrate. At an aerator detention time of five hours, a COD:N ratio of 60:1 and a COD:P ratio of 140:1, good purification efficiency was maintained. No serious disruption resulted from the quantitative shock loads applied to the system, and no NH_4^+ was detected in the effluent. Sludge wastage was approximately 32%.

In all continuous flow operations, as shown in Figure 27, two aerators were used for the regeneration and recycling of sludge. These were operated as batch systems. Each twelve hours, sludge was withdrawn from the clarifier underflow, diluted with effluent to a constant concentration (to maintain X_R constant), and a portion was wasted. The remainder was aerated for twelve hours in the presence of NH_4^+ at a concentration based on the amount of COD fed to the feeding aerator during the preceding twelve hours. The regenerated sludge was then pumped to the aerated holding tank, from which it was recycled at a constant rate during the succeeding twelve hours while a new batch of sludge was being regenerated. Preliminary studies of required regeneration time indicated that the detention time in the endogenous aerator could be reduced.

Figure 28 shows typical data for effluent analyses during continuous flow operation of the oxidative assimilation process with synthetic waste containing acetic acid as the sole carbon source (58) The hatched area represents the synthesis of storage products. The synthetic waste contained no nitrogen, and the ratio of COD:N used as a basis for nitrogen supplementation in the endogenous aerator was 70:1. The efficiency of solids removal in the clarifier was approximately 96%. These data demonstrate the potential of the proposed process for treatment of nitrogen-deficient industrial wastes.

There is some obvious similarity in the flow sheets for the oxidative assimilation process and the biosorption, or contact stabilization process, and it might be possible in some cases to convert biosorption plants to operation as oxidative assimilation processes. Both types of treatment involve dual aeration tanks; however, relative detention times for the first and second aeration periods may be quite different. The

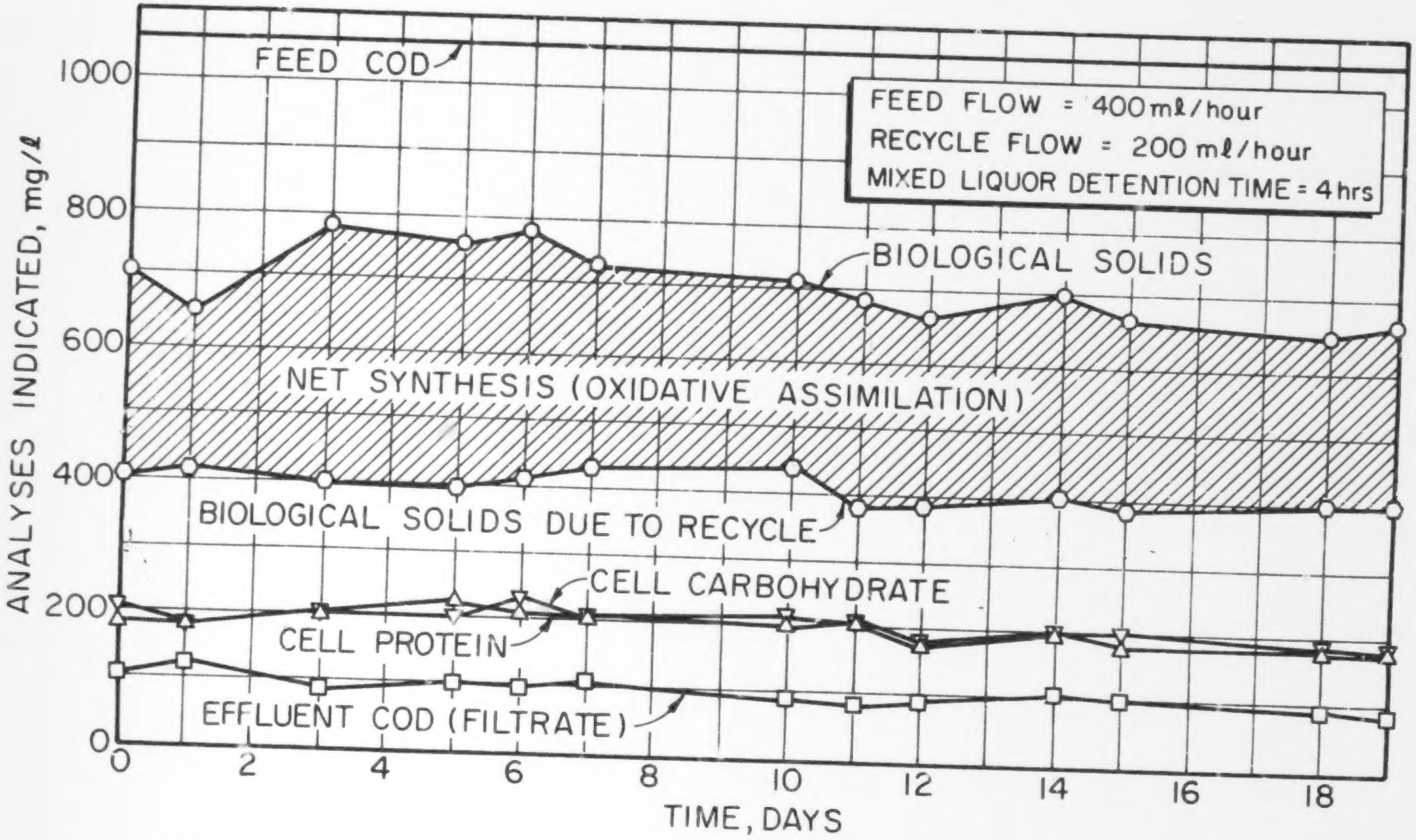

Figure 28. Operational data for treatment of nitrogen-free acetic acid waste by continuous oxidative assimilation process with nitrogen supplementation in the endogenous aerator at a COD:N ratio of 70:1 (58).

fundamental difference between the oxidative assimilation process and biosorption lies in the point of addition of nitrogen. In the biosorption process, nitrogen is added to the waste before the carbon source is removed, and in oxidative assimilation, the carbon source is removed before nitrogen is added. In the latter process, nitrogen is added only to the sludge, not to the waste, and only to that portion of the sludge which is to be recycled. In the biosorption process, nitrogen cannot be used efficiently and much of the added nitrogen is likely to be lost in the effluent. This occurs because, as our studies have shown, at very high solids:COD ratios, such as those used for biosorption, the carbon source is removed and stored in the cells before synthesis of nitrogen-containing compounds occurs, even if nitrogen is present. Thus, it is likely that most biosorption plants actually remove carbon source largely by oxidative assimilation and that much of the added nitrogen is wasted because it is made available at a time when the cells cannot utilize it efficiently. The oxidative assimilation process eliminates leakage of nitrogen in the effluent and allows a considerable reduction in the cost of nitrogen supplementation.

Summary and Conclusions

Studies of the kinetics of substrate removal showed that at high initial ratios of biological solids to COD, nitrogen utilization and cell replication occur primarily after the COD has been removed. In these systems COD removal is largely accomplished, even in the presence of nitrogen, by oxidative assimilation. The carbon source is removed and converted to internal storage products, which are then used for synthesis of the nitrogen-containing compounds necessary for formation of new cells. Since carbon source assimilation and nitrogen utilization occur sequentially in the cell, it is possible to separate these processes physically in a new flow sheet proposed for treatment of nitrogen-deficient industrial wastes.

The proposed flow sheet, shown in Figure 26, utilizes two aerators. In the first, or feeding, aerator, nitrogen-free waste is treated by oxidative assimilation. A nitrogen-free effluent is produced since no nitrogen is added to the waste. The sludge to be recycled is regenerated by addition of ammonia-nitrogen in a second (endogenous) aerator. A considerable savings in the cost of nitrogen supplementation is realized, since a COD:N ratio much higher than that used in traditional methods of treatment can be employed. Laboratory-scale pilot plant studies using several synthetic wastes and a sugar refinery waste have demonstrated the feasibility of the proposed continuous oxidative assimilation process for treatment of nitrogen-deficient industrial wastes.

VIII

EXTENDED AERATION PROCESS

The present document is concerned with conceptual principles relevant to the purification of wastes, i.e., with metabolic occurrences taking place largely in the aeration chamber. It is not our intention to cover the topic of sludge disposal, although ultimate disposal of organic or biological sludges is a vital concern in the water pollution control field. Most engineers are, however, aware of the "extended aeration" process or the "total oxidation" process, in which biological treatment of the incoming waste is effected via an activated sludge and, concurrently with this function, the activated sludge is subjected to aerobic digestion--the end result supposedly being net conversion to CO_2 and H_2O of all of the organic matter originally in the waste stream or at least of that portion of it which is biologically treatable. Concurrent waste water purification and aerobic digestion are accomplished by returning all sludge to the aeration chamber. The organic matter in the sludge is subject to wet biochemical combustion, and is dissipated as CO_2 and H_2O just as when it is ultimately disposed of by chemical combustion or anaerobic digestion and burning of methane. Thus, if the approach to ultimate disposal is recycle of the waste organic carbon by converting it to innocuous inorganic carbon, CO_2, combustion, i.e., consumption of oxygen, is inevitable. Since in an activated sludge process the oxygenation energy which is supplied to the system is much in excess of the metabolic requirement, the attractiveness of accomplishing both treatment of the waste and wet combustion of the sludge developed during treatment can be readily appreciated.

This unique modification of the activated sludge process was suggested by Porges and his co-workers as a consequence of their research on the biological treatment of dairy wastes (63). It was suggested that if the biological solids were continually returned to the aeration chamber (no wasting of excess solids), endogenous respiration would tend to decrease the solids concentration by about the same amount that the new incoming carbon source in the waste would tend to increase it, and eventually a state of equilibrium might be approached at which there would be no net solids accumulation. Thus, the dual function of biological waste water purification and sludge digestion (aerobic) could be accomplished concurrently and without recourse to disposal of any biologically produced sludge.

At the time the process was suggested, the term "endogenous respiration" was subject to various interpretations; it was at that time a rather new term to the field of biological waste treatment. Interpreted solely as the intra-cellular oxidation of organic carbon, one could argue against the theoretical possibility of total oxidation, since it is obvious that a given microbial cell cannot oxidize itself totally to inorganic carbon. Also, it could be reasoned that if one cell died, not all of its organic carbon might serve as available sources of carbon for another species; thus the validity of the concept was suspected. In any event, the suggested process modification precipitated much interest and engendered much research activity by other investigators (e.g., 64, 65). Two general conclusions emanated from these investigations. First, it was concluded that certain component parts of the biological cells synthesized in the purification phase were nonmetabolizable. Thus there was a fraction of organic matter which was inert from two points of view: it was dead organic matter, therefore incapable of removing substrate, and it could not serve as food material for live cells. As time went on and this material was retained in the system, the bio-mass would thus consist of an ever-increasing inert fraction, and eventually the system must fail. The other conclusion, which was somewhat related to the first, was that a condition of biological solids equilibrium could not be attained, i.e., the biological solids concentration would continually build up. It was generally felt that biological solids had to be wasted, or the system could not function. The concept of total oxidation was generally concluded to be theoretically untenable, and biologically unsound. Regardless of these conclusions, many extended aeration plants have been built, and operational experience has varied regarding their worth and the need to waste biological sludge periodically.

Recent Investigations of Extended Aeration

A few years ago, based upon some of the experimental observations made during investigations pertinent to our research on "kinetics and mechanism of activated sludge processes" there was, we believed, some cause to doubt the finality of the previous conclusions concerning the possibility of total oxidation. Also, the paradoxical situation in the field wherein extended aeration plants were being installed in opposition, as it were, to the conclusions of some engineering researchers, certainly seemed to require some reconciliation. Also, there were no definitive data on how long an extended aeration activated sludge might continue to function until it experienced failure, i.e., until it no longer could function as an effective means of removing soluble organic matter from waste waters. It was reasoned that even if, after a year or so, some sludge had to be wasted and new bio-mass developed, the process would still be advantageous since continuous operation of a sludge treating and disposal facility would not be necessary. Thus it might have considerable utility and applicability to treatment of various industrial wastes. All of these considerations contributed to a decision to enter upon long-term investigations of the process.

In order to test the theoretical possibility of attaining total oxidation, i.e., to determine whether continual buildup of biological solids containing an ever-increasing fraction of biologically inert solids occurred, it was necessary to devise a laboratory pilot plant scheme which could guarantee that all biological solids were retained in the system. The problem of inadvertent loss of biological solids in the effluent had been experienced by other workers, and was one of the factors which originally caused us to doubt the validity of some of the conclusions which had been drawn from their results. The pilot plant and operational techniques we employed are described in detail in the research literature (66-68). Briefly, positive control over retention of biological solids in the pilot plant system was maintained by centrifuging all effluent from the clarifier and returning the solids to the aeration tank. Thus, even at times when solids could have exited a more usual pilot plant, they did not exit the one which we operated. A small amount of solids was removed for analysis. Such removal amounted to no more than 0.2 per cent of the bio-mass.

The organic feed to the unit consisted of a "synthetic waste" in which the carbon source was a soluble carbohydrate (glucose). It is important to consider why a simple soluble compound such as glucose was employed, since it has been the experience of researchers that some in the pollution control field have a tendency to discount work with "synthetic wastes." Some of the pros and cons of this and other approaches to research in the environmental field have recently been presented by Gaudy (69). In the present instance, it is important to point out that the crux of the question concerning the theoretical unsoundness of the total oxidation process was the supposed non-biodegradability of various fractions of the microbial cells of which the activated sludge is composed, e.g., the extra-cellular polysaccharide slime layer (64). It is essential, then, to make sure that the activated sludge one is examining does consist wholly of microbial cells. Obviously, entrapped organics such as coffee grounds, and inorganics such as silt, etc., are relatively or completely inert to aerobic biological treatment. It is emphasized that the previous conclusion that the principle of total oxidation was unsound was based upon experimentations employing simple soluble carbon sources, e.g., acetic acid (64), and glucose (65). While we are in disagreement with the conclusions of these workers, we agree that their use of simple soluble carbon sources was wholly justified.

If one retained all of the biological solids in the system, and if there were a permanently inactive organic fraction, the biological solids concentration should continue to increase; i.e., one would expect continual accumulation of biological solids. Also, if bacterial polysaccharide in the cell capsule or slime layer were biologically inert, the carbohydrate content of the sludge would gradually increase. The ability of the activated sludge to remove the carbon source in the waste would gradually decrease. Another indication of an increasing inactive fraction of the sludge might be a decrease in basal rate of respiration (O_2 uptake rate per unit of sludge under endogenous conditions, i.e., in the absence of an extra-cellular source of carbon). These and other aspects were investigated during a three-year period of continuous pilot plant operation with centrifugation and total solids retention.

It was found that the biological solids concentration did not reach an equilibrium, or steady, value in which "endogenous respiration" balanced new synthesis. However, neither did the biological solids concentration continually increase. There were periods of solids increase and periods of decreasing biological solids concentration. During a decreasing cycle in biological solids concentration, the organic matter (COD) in the effluent did not rise, as would have been the case if some cells had undergone lysis and the organic matter thus released were not metabolized by the remaining intact cells. One of the most dramatic downward cycles occurred during the latter part of the first year of operation (67). Figure 29 shows the operational behavior of the system during this cycle. From day 285 to 307, the biological solids concentration decreased from approximately 8500 to 2400 mg/l. Despite this rather sharp drop, the filtrate COD (effluent) did not rise appreciably. In fact, there was only a slightly perceptible rise with a rapid return to the previous level. The COD of the synthetic waste was approximately 530 mg/l (500 mg/l glucose), and the filtrate COD was for the most part less than 40 mg/l except for the short-lived transient as the solids began to decrease, when it rose to approximately 100 mg/l.

Since solids were not wasted (purposely or accidentally), and since they decreased without any concomitant rise in COD, and since no external operational changes were imposed upon the system, this downward cycle in biological solids was a natural one brought about by the system itself. During such a decreasing cycle in biological solids concentration with no rise in effluent COD, the most apparent explanation is that a portion of the sludge (microbial population) is serving as a source of carbon (substrate) for another portion of the microbial population. In short, total oxidation of the organic material in the waste must occur by a combination of the reactions shown in equations 3 and 13, with recycling of the carbon through the latter. It cannot be expected that the amount of new growth on the incoming substrate will always be balanced by cannibalism of the past growth so as to produce a steady or equilibrium level of solids. The predominance of species is constantly shifting, and it is necessary that the ecology of the system be such that some of the cells extant in the system act as food material for other cells which play the role of feeders. Neither the periodicity nor the extent of cyclic accumulation and de-accumulation of biological solids can be predicted, but the results shown in Figure 29 surely attest to the occurrence of such periods.

It should also be pointed out that during more than 1000 days of operation there was no evidence for the buildup of carbohydrate content in the sludge. Samples taken for measurement of protein and carbohydrate content of the sludge close to the end of the pilot plant operation [e.g., see Figure 5, reference (68)] show that the carbohydrate content was approximately 20%, which represents a normal value for microbial cells.

Many batch feeding tests to determine the substrate removal capability of the sludge were periodically performed (67, 68). Figure 30 shows

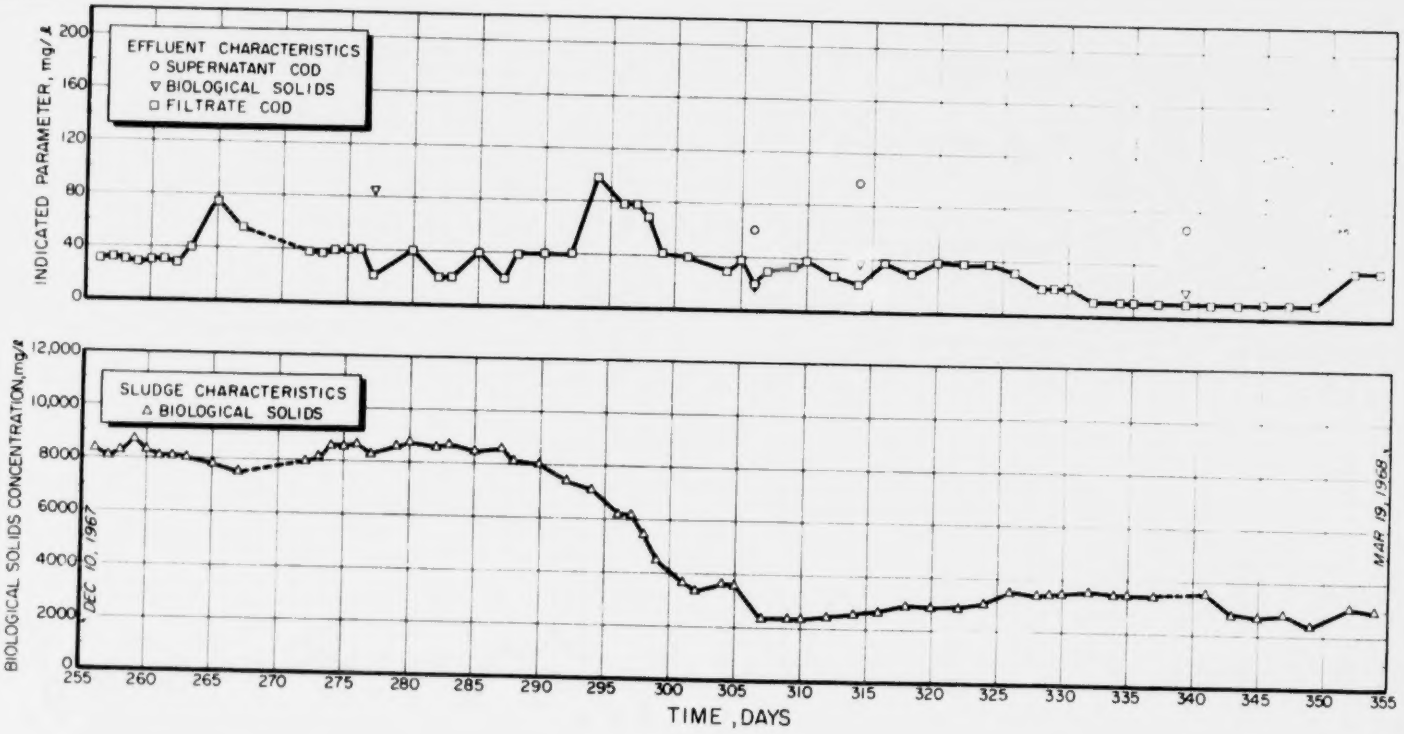

Figure 29. Performance data for a continuous flow extended aeration activated sludge in a laboratory pilot plant operated with total solids recycle, showing a period of accelerated autodigestion (67).

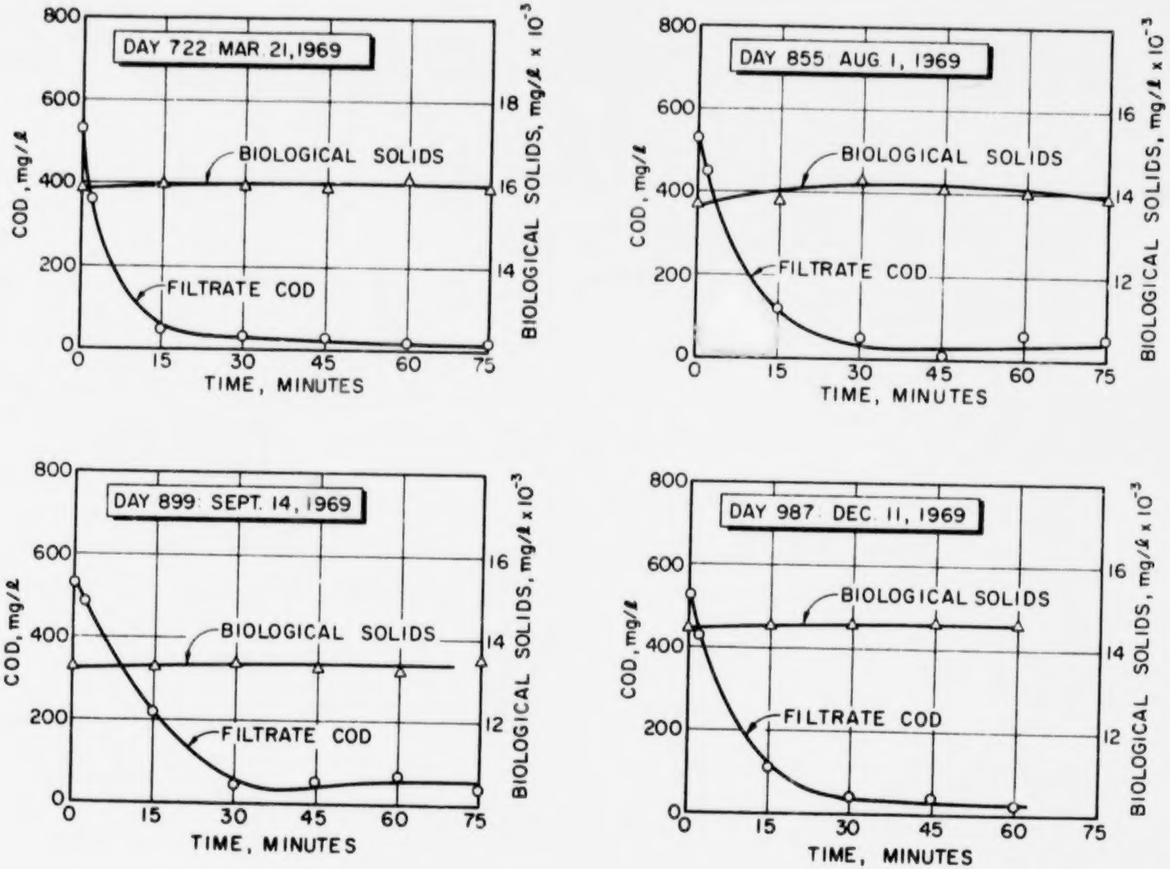

Figure 30. COD removal capability of extended aeration activated sludge after the number of days of continuous operation with total solids recycle indicated on each graph (68).

the results of four such experiments. Even after 987 days of operation with no sludge being wasted, the system removed the daily feed, added as a slug dose, within thirty minutes. Considering the fact that the retention time in the aeration chamber was sixteen hours under continuous flow conditions, the capacity for substrate removal was far in excess of that required even after nearly three years of aging. This would not have been possible had the sludge not possessed the ecological and biochemical capacity for self-renewal, i.e., internal feeding. It is important to point out that the basal respiration rate (endogenous O_2 uptake rate; mg O_2/hr/gm sludge) was considerably lower than for a younger microbial population. This is an indication that the sludge was somewhat less active than young cells on the basis of unit weight. For example, the endogenous uptake rate for this sludge appeared to level off between 1 and 2 mg/hr/gm, as is shown in Figure 31. Young populations freshly grown from a small inoculum of cells from the aeration chamber manifested basal respiration rates approximately ten times higher. The rising trend in the values between days 300 and 350 occurred immediately after the period of accelerated autodigestion (decline in solids) shown in Figure 29. During and after such a period of declining biological solids concentration, there is a larger fraction of younger cells in the sludge because of the higher substrate concentration (i.e., the cannibalized cells), and as a consequence of this physiological condition, higher endogenous O_2 uptake rates were manifested.

The facts that biological solids did not continually accumulate, that the carbohydrate content did not attain abnormal proportions, and that the system continued to deliver excellent substrate removal efficiency after three years of continuous operation with total cell recycle, bring us to the conclusion that the conceptual principle of the total oxidation or extended aeration process is not theoretically unsound, and work designed to gain a more definitive insight into the metabolism of cellular components is continuing.

In addition to the long-term studies just described, we have run other experiments in closed batch systems which demonstrate that the total oxidation of the bio-mass is possible. Results of one such experiment are shown in Figure 32 (62). In this experiment, a relatively low initial concentration of acclimated sludge was fed 2280 mg/l glycerol COD and various parameters were monitored during the ensuing growth and "endogenous" periods. The top portion of the figure shows changes in protein and carbohydrate content of the sludge as well as ammonia nitrogen, which was the only source of nitrogen in the system. The lower portion of the figure is particularly important for present considerations, because it can be seen that the concentration of biological solids, for all practical purposes, returned in the prolonged endogenous phase to the initial concentration; thus there was no "permanent" synthesis product. Essentially total oxidation of the assimilated carbon was effected in this closed batch system.

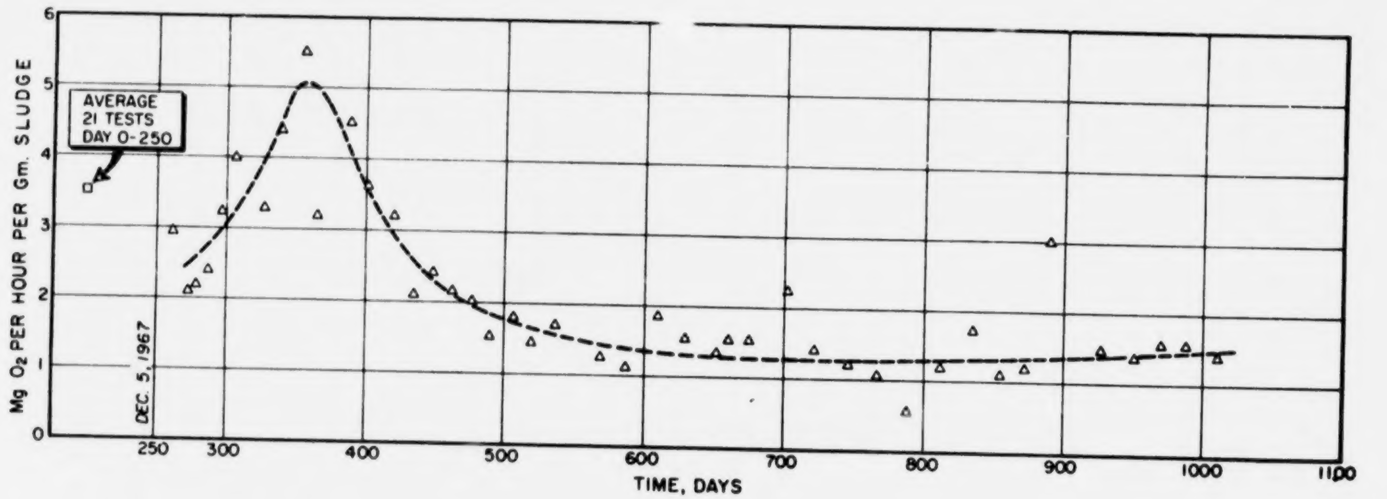

Figure 31. Effect of aging on endogenous oxygen uptake by an extended aeration activated sludge during long-term pilot plant studies employing total solids recycle (68).

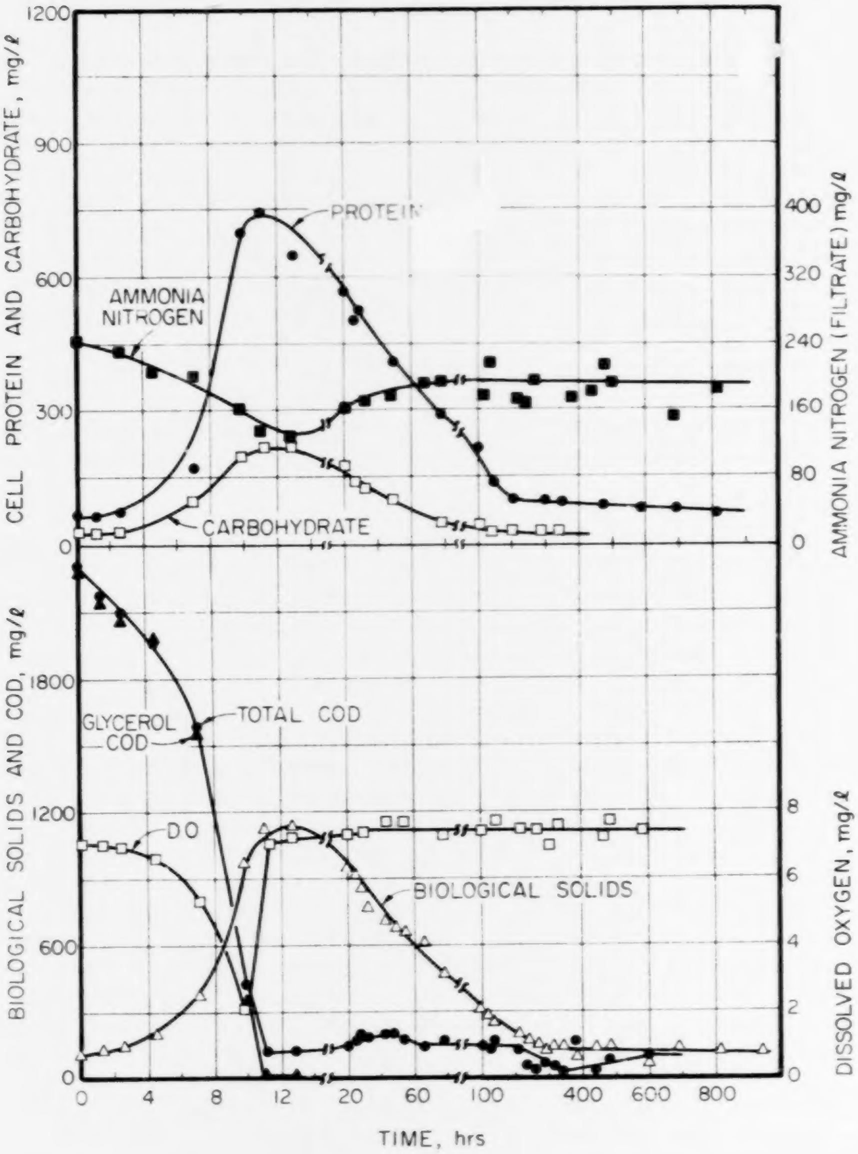

Figure 32. Total oxidation, during the endogenous phase, of the biological solids synthesized during the substrate removal phase in a batch system using a heterogeneous population developed from a sewage seed (62).

Engineering Control of Total Oxidation

Because the total oxidation process has been shown to be conceptually sound does not automatically confer upon it practicability or engineering feasibility, but removal of the onus of conceptual unsoundness from the process allows engineering considerations to bring theory to practical fruition. No one can predict how long a total cell recycle system will continue to accumulate biological solids before the ecological (predator-prey) relationship is such as to initiate a period of accelerated autodigestion. No one can predict the extent of the ensuing decrease in biological solids concentration. All that can be said is that if the cells can be retained in the system, periods of deaccumulation will occur. At times, solids may accumulate to very high concentrations, thus making retention difficult with currently employed field procedures. It seemed essential, therefore, to explore engineering possibilities for attaining control of biological solids concentration, especially ways to initiate the autodigestive process.

When cells are totally oxidized, all portions of the cell must be converted to simple inorganic compounds which include CO_2, H_2O, NH_3, and small amounts of inorganic salts. Even for readily metabolized compounds, approximately five passes through the food chain would be required for reduction of the original amount of carbon source to the desired level of purification. This is based on an assumption of 50% average yield for metabolism, i.e., the assumption that approximately 50% of a substrate is oxidized and 50% converted to new cell material. When the newly formed cell is oxidized by other cells, if the yield is again 50%, the amount of original carbon source still present as cell material is 25%, and each succeeding pass reduces this by a factor of one-half. For components of the cell which are metabolized with difficulty, long periods of time may be required for complete oxidation. Initiation of oxidation of cellular components may depend upon access to them by other cells which are capable of utilizing them. Some of the carbon in a microbial cell is in soluble form, but a large part of it is incorporated in larger insoluble macromolecules in the cell membrane, cell wall, and extra-cellular capsular material, e.g., polysaccharide, and in the proteins and nucleic acids within the cell. Some of the predatory activity which occurs in the sludge involves the ingestion of particulate organic matter, e.g., ingestion of some species of bacteria by some species of protozoa and other predatory microorganisms, and other activity involves the use of organic carbon of some species of bacteria by other species of bacteria. This organic substrate must be transported into the cell if it is to be metabolized. Therefore, the first and probably the most difficult metabolic task is the breaking down of the organic macromolecules, i.e., their hydrolysis to smaller fragments which can be taken into the cell. It is indeed a very complex ecology which is needed for all of these steps to occur. Our results have demonstrated that they do occur; i.e., these biological occurrences are not theoretically impossible. However, one cannot "engineer" the ecology of the system, at least not at present. We are continuing studies to gain insight into understanding this useful natural occurrence (a prerequisite to attempts to engineer the natural ecosystem).

There are ways to approach control of the system other than learning to exert environmental control of the ecosystem. One such channel, which we felt warranted investigation, was to combine aspects of chemical and biological digestion of the sludge. One of the mildest digestive chemical operations and the easiest to perform is hydrolysis, the preparatory step to biological digestion. If one provided such an engineering assist to the system, the autodigestive process might be subjected to direct engineering control. With the hydrolysis portion of the digestion process subject to engineering manipulation, the heterogeneous microbial population might be better able to metabolize that portion of the sludge which had been hydrolyzed and returned to the aeration compartment. This process, like the one discussed in the previous chapter, would represent a rather radical departure from current practice or concept, but it seemed a plausible one in view of our previous findings and was consistent with the general aim and need for innovation in the pollution control field. Experiments were initiated to test this concept.

Figure 33 shows the course of COD removal when hydrolyzed sludge was fed to the extended aeration pilot plant in a batch experiment similar to those shown in Figure 30 in which the synthetic waste (glucose) was fed. It is seen that the hydrolyzed sludge was metabolized rather rapidly (compare with Figure 30) by the extended aeration activated sludge. These hydrolyzed cells had been rendered essentially soluble by acid hydrolysis (pH = 1) for five hours at 15 psi (1.05 kg/sq m) and 121°C. The hydrolyzed sludge was neutralized to pH 7 and fed as a slug dose to the pilot plant.

On the basis of such results, we have proposed the modification of the extended aeration process embodying the concept of the hydrolytic assist shown in Figure 34 (68). The type of extended aeration process shown is the compartmentalized aeration chamber-clarifier with internal sludge recycle (dotted arrow). The lower portion of the flow diagram shows the hydrolysis process. This portion of the plant can be considered as an aid to aerobic sludge digestion in which a portion of the total return sludge is solubilized before return. In the extended aeration process, the return sludge itself is really a waste stream being fed to the system like the incoming waste. The modification proposed in Figure 34 simply prepares a portion of the sludge so that it becomes a more readily available substrate, thus allowing some engineering control over the biological process. With the chemical hydrolysis modification, one is provided with a sludge disposal aid which need be used only when required, and for which there is no long startup time. The hydrolysate could be stored without neutralization, thus preventing odors, etc., and neutralized as it was channeled back to the aeration chamber at a rate controlled and selected by the operator. Such a system uses both chemical and biological methods of sludge disposal to their best advantage.

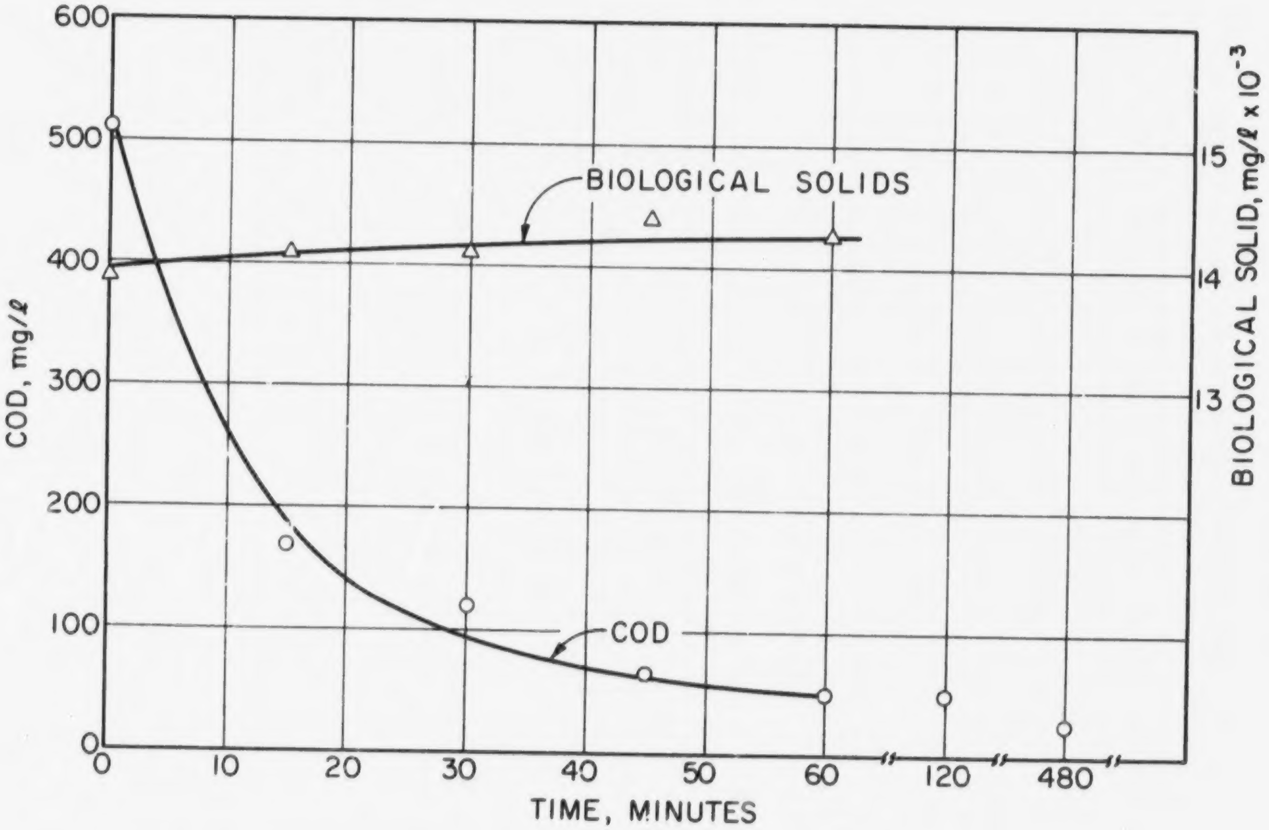

Figure 33. Metabolism of hydrolyzed activated sludge by an extended aeration activated sludge (68).

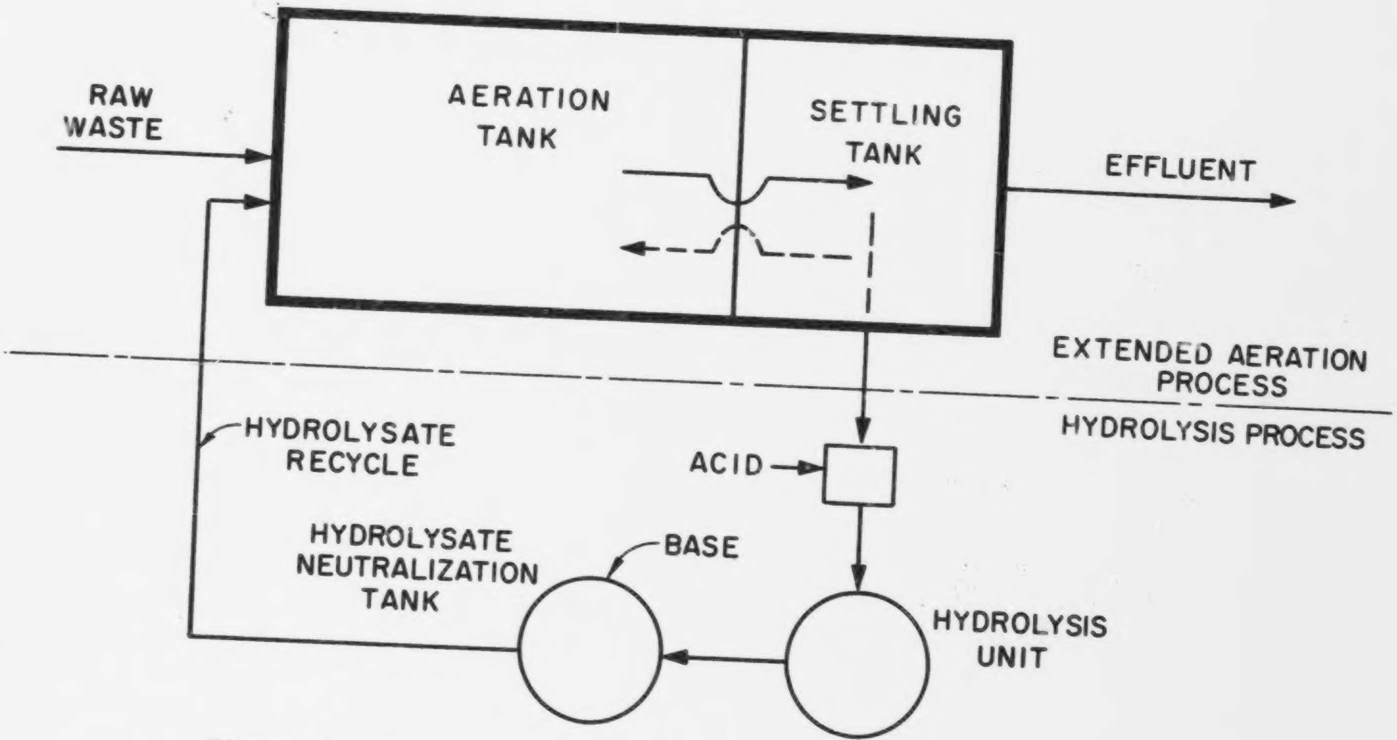
RAW WASTE
AERATION TANK
SETTLING TANK
EFFLUENT
EXTENDED AERATION PROCESS
HYDROLYSIS PROCESS
HYDROLYSATE RECYCLE
ACID
HYDROLYSATE NEUTRALIZATION TANK
BASE
HYDROLYSIS UNIT

Figure 34. Proposed extended aeration activated sludge process incorporating chemical hydrolysis for control of sludge concentration (68).

It performs chemically a function which is biologically difficult, and performs biologically a function which is difficult and costly to perform chemically. Additional investigational work is warranted and is in progress; to date we have successfully operated the bench scale pilot plant with periodic sludge hydrolysis, and shortly a report on these operational experiences will be prepared for publication in the technological literature.

It may be noted in Figure 34 that "raw waste" flows into the aeration tank. The figure was drawn in this way because it is generally the case that where extended aeration is employed, an attempt is made to require the process to perform the function of primary treatment as well as secondary treatment and sludge disposal, i.e., there is no primary settling tank. In the "traditional" flow sheet for treatment of municipal wastes, secondary and primary sludges are combined and channeled to anaerobic digestion, so it is probably natural that with the extended aeration process, both types of sludge would be expected to be combined for aerobic digestion in the aerator; i.e., it is natural to look upon the non-soluble organic material in primary sludge as a substrate just as the return sludge, the microorganisms, are considered as a particulate substrate in the extended aeration process. However, one does not expect such panaceas with other activated sludge processes, and there is no real justification for expecting an extended aeration activated sludge to totally oxidize primary sludge. The probability is that it will not. The extended aeration process is best considered as a secondary biological treatment process. Also, it is ideal for waste containing primarily dissolved organic matter. In this regard, it is better suited to treatment of some industrial wastes than to municipal sewage (where it has been largely employed). However, with the hydrolytic assist process we are recommending, it may be possible that some particulate organic matter in the raw waste which becomes entrapped in the microbial sludge (and, indeed, is partially metabolizable) can be hydrolyzed under the same conditions which solubilize the cells. In this case, some of the organic matter which normally would be removed by primary settling is treatable "chemo-biologically." Some organic compounds in "primary" municipal sludge can undoubtedly serve as excellent microbial substrates with or without chemical hydrolysis. However, constituents such as coffee grounds, for example, would be difficult to hydrolyze chemically or to metabolize aerobically; thus for municipal sewage some organic residue should be expected, i.e., total oxidation of such sludges may not be possible. However, the residual organic sludge which would have to be disposed of should be low in amount and, by virtue of its resistance to both chemical hydrolysis and biological attack, should be relatively non-putrescible. Thus, addition of the hydrolytic assist should allow the extended aeration process to be very usefully employed even for municipal wastes. The non-hydrolyzed sludge would have to be separated from the hydrolysate prior to channeling the hydrolysate to the aeration chamber. Removal of finely divided suspended organic matter in the hydrolysate could be accomplished with chemical addition (coagulation, flocculation, and settling) following the hydrolysis unit. This would also remove finely divided inorganic matter which may pass the grit chamber and be entrapped in the sludge. It should be apparent that suspended matter which is initially inert, i.e., non-biodegradable, should remain so after passage through the aeration chamber, after possible entrapment in the bio-mass and after hydrolysis. Indeed, the hydrolysis modification including chemical precipitation of permanently inert material originally in the waste stream provides a useful way of separating the pollutional (treatable) from the non-pollutional material in the waste.

Summary

In summary, based upon long-term and thorough studies, we are able to conclude that the extended aeration or total oxidation process is not conceptually unsound. The proposed modification of the process embodying the hydrolytic assist offers an effective way to combine ideal features of chemical and biological treatment for concurrent treatment of the waste water and essentially total aerobic digestion of sludge developed in the process.

IX

SUMMARY AND RECOMMENDATIONS

At the outset it was emphasized that this document was not intended to serve as a manual for design and operation of activated sludge processes, but as a conceptual guide for updating and unifying mechanistic and kinetic principles which bear strongly on design and operation. The real aim was to help close the so-called "gap" between research and practice, and this we have attempted to do by relating, largely in readily understood engineering terms, many of the findings of our fundamental research effort and the concepts which have been developed over the past decade. While our research has been, and in large measure will always be, designed to gain better basic understanding of biological processes, our ultimate aim is the useful application of the information. Thus, throughout this document, presentation and discussion of conceptual principles have been accompanied by embodiment of these principles in recommended procedures, models, process innovations, etc., for the consideration of the profession.

Beginning with generalized concepts of exertion of biochemical oxygen demand and use of ΔCOD as the measure of biologically treatable organic material, through discussion of the stoichiometry of the treatment process and development of growth kinetics, incorporation of the kinetic relationships into a relatively simple model for description of completely mixed growth reactors and, finally development of a modified kinetic model specifically for the activated sludge process, the approach has been to foster better understanding and to provide recommendations leading to engineering control over the biological treatment process. The model finally evolved in Chapter V, equations 43 through 47, embodies the concepts developed in the previous chapters, and represents an engineering modification of basic kinetic theory. The modification, i.e., holding X_R rather than c as a system constant, complicates the kinetic equations somewhat but allows for practical attainment of greater steadiness in X and S. The goal in biological treatment is attainment of steady, reliable delivery of the needed degree of process efficiency, and the recommended model for design and operation should foster this aim. The graphs of Figure 18 or 19 can be used as a rough guide to design, although the engineer should ideally determine values of the biological constants μ_{max}, K_s, and Y for the particular waste in question. In computing the graphs shown in Figures 18 and 19, values of the physical constants chosen were X_R = 10,000 mg/l, and α = 0.25, which are reasonably selected values; however, the design engineer can, with the model, select values of his own choice and can test the effect of his selection on $\bar{X}$ and $\bar{S}$.

It has been emphasized that if one desires a reasonable biological engineering model for a process, he must provide as best he can in his design the facilities for control of the parameters included in the model.

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Thus, means of controlling X_R nearly constant at the selected value should be provided. This requires the incorporation of a sludge consistency tank, i.e., a revision of the "traditional" flow sheet. The sludge consistency tank is somewhat analogous to the make-up tank for chemical dosage used in chemical treatment. The sludge consistency tank is one in which the biochemical "dosage" to the activated sludge aeration tank is prepared. Also, some attempt to even out fluctuations in F and S_i may be required either on the line or at the treatment plant. The use of an equalization basin to accomplish this is recommended where practicable. While equalization basins have been used in some installations, their use is a departure from the general practice. Thus, the newer concepts we have introduced for the "normal" activated sludge process, involve some changes in the usual or traditional flow diagram. These changes are intended to help the system approach the "steady state" assumption of the kinetic model.

One should realize that regardless of which processes one employs in the treatment of waste waters--chemical, physical, or biological, or any combination thereof--an effort must be made to hold the system steady or to protect against drastic changes in the operational environment. After all, chemical-physical treatment plants are just as likely to undergo stress leading to fluctuations in efficiency, due to changes in S_i, F, or the amount of coagulant or adsorbent chemicals added, as are biological treatment plants. For example, one would not think of chemical treatment without control of chemical dosage. In like manner, one should also provide control of the biological dosage, i.e., X_R, or, in any event, should not let X_R fall below a selected level. Thus, building in the engineering control to make the process function in accord with the preconceived notion (some sort of design model) is not incongruous with accepted engineering practice. It is probably more true of biological treatment systems than of chemical-physical ones, that the system possesses an internal resistance and responsiveness to external change. For example, the agents for removal of organic matter (the microorganisms) are self-producing, and do essentially "rise to the occasion" of high organic loadings, i.e., there is a self-correcting feature in biological systems which, if not overloaded by abusive external change, serves to protect the system and to provide some internal regulation and restoration of efficiency when disruption does occur.

While the material in Chapters II through VI can be said to pertain to general understanding of activated sludge processes, and in these chapters some engineering innovations which depart from traditional practice were introduced, in Chapters VII and VIII more radical departures from the norm were covered. In Chapter VII, the effect of nitrogen as limiting nutrient and the effect of biological solids concentration on the predominant biological mechanism of substrate removal were delineated. The new process modification which we have termed "continuous oxidative assimilation," has been demonstrated to be feasible, at least in laboratory scale pilot plant studies. The process has unique possibilities for nitrogen-deficient industrial wastes and, in addition, should serve to demonstrate that some of the older concepts and nutritional guidelines are by no means inviolable.

It should perhaps be pointed out that the finding that COD may be removed primarily by oxidative assimilation in batch or continuous flow systems containing high concentrations of biological solids does not negate the use of kinetic models for nitrogen-containing wastes, based on equations for microbial growth, such as those presented in Chapter V. The continuous oxidative assimilation system is one in which substrate removal and cell replication are separated artificially. The cells are forced to carry out oxidative assimilation and protein synthesis in a completely sequential manner by addition of carbon source and nitrogen source to separate aerators. The sequential synthesis of storage products and nitrogen-containing cellular components observed in batch systems, even in the presence of nitrogen, would undoubtedly be observed in activated sludge systems closely resembling plug flow reactors. It is not possible to determine whether such sequential synthesis occurs to an appreciable degree in completely mixed reactors with high solids, i.e., in reactors employing cell recycle, because one cannot examine synthesis in individual cells and it is the newly recycled cells which would be most likely to carry out oxidative assimilation. We consider it likely that oxidative assimilation does occur to some degree in these reactors. Activated sludge is not completely comparable to the usual growing systems of the microbiologist or fermentation engineer. In the first place, as was brought out in Chapter V, the recycling of cells forces a very slow rate of replication, a rate which is only a fraction of the system dilution rate. Secondly, the recycled cells have an immediate past history of existence in an environment essentially devoid of carbon source (the clarifier). Because of these two reasons, as well as the general starvation condition of the system, these cells cannot be classified as physiologically young, actively growing microorganisms. Therefore, when sludge is returned to the aerator, the cells may remove carbon source initially by oxidative assimilation, possibly to an extent depending primarily upon the retention time in the secondary clarifier. However, in systems in which the solids:COD ratio is high, i.e., systems in which $1 + \alpha - \alpha c$ is small, the doubling time for the population may extend over several aeration periods. Therefore, in high solids systems, a very long period of time is available to the cell to prepare for replication. For maintenance of a steady state in X in systems with recycle, it is only necessary that cells replicate at a very slow rate or, in other words, that a relatively small percentage of the cells in the aerator replicate during one detention period. Since batch studies have shown that in high solids systems the rate of increase in X (measured in units of mass, not cell numbers) is not very different for cells which are utilizing carbon source in the presence of nitrogen or in its absence, the occurrence of oxidative assimilation during part of the cell division cycle of individual cells should have no noticeable effect on the kinetics of the system.

Chapter VIII deals with the extended aeration process, which incorporates the important aspects of sludge disposal and secondary treatment of the waste. It was shown that this process is not biologically unsound and a new modification of the process (the hydrolytic assist), which allows engineering control of the process, was proposed. This modification is a union of chemical and biological treatment which can provide a useful alternative or replacement for anaerobic digestion and/or incineration of waste organic matter from treatment plants. It accomplishes the same thing as these processes, i.e., conversion of organic carbon to inorganic carbon. In the long range, any method of disposal which provides for recycle of carbon via conversion of waste organic carbon to CO_2 for ultimate reconversion via the photosynthetic process, may become obsolete, since this process is a slow recycle and the organic carbon in the waste may need to be recycled for man's use by more direct means; e.g., it may eventually be a usable byproduct of water rejuvenation or a unique raw material for production of useful commodities. However, in the near future, the conversion of these waste organic sludges to CO_2 by whatever means, and the dissipation of CO_2 into the atmosphere represents, if not the ultimate, the "penultimate" sludge disposal method. This is by any means a combustion process, and the chemically-assisted wet aerobic biological combustion we have herein recommended for consideration of the profession should be accomplishable without the remedial and often costly expedients to abate air pollution problems usually required with other means of disposal.

Figure 35 shows a possible flow diagram for an activated sludge plant which embodies many of the conceptual principles discussed in this report. Since there are really no standard wastes, there is no standard way of treating all wastes. Accordingly, all activated sludge process schemes should not be expected to look the same, but each should be tailored to the unique requirements of the particular situation. Uniqueness and engineering innovation rather than standardization should be the rule, provided that the engineer can defend the reasonableness of the concepts and has performed pre-design investigations to justify omission or addition of any particular unit operation or process in his overall plant design. While we feel that the flow diagram shown in Figure 35 may offer considerable improvement over the more or less standardized flow diagrams which have traditionally become associated with activated sludge processes, we do not offer it as a new standard flow sheet but as the general embodiment of many of the conceptual principles we have developed in the previous chapters. It is in a design sense, a tentative "diagrammatic" conclusion which seems a meaningful result of the transference of conceptual principles and research findings to functional design. It is readily appreciated that this flow diagram differs from the traditional representation of an activated sludge plant in a number of ways.

The essential units are shown in solid lines and those which may or may not be needed, depending upon the particular situation, are shown in broken lines; the surge basin falls into the latter category. If the waste flow and strength can be regulated within the guideline limits previously given without a surge basin, and if drastic changes in the

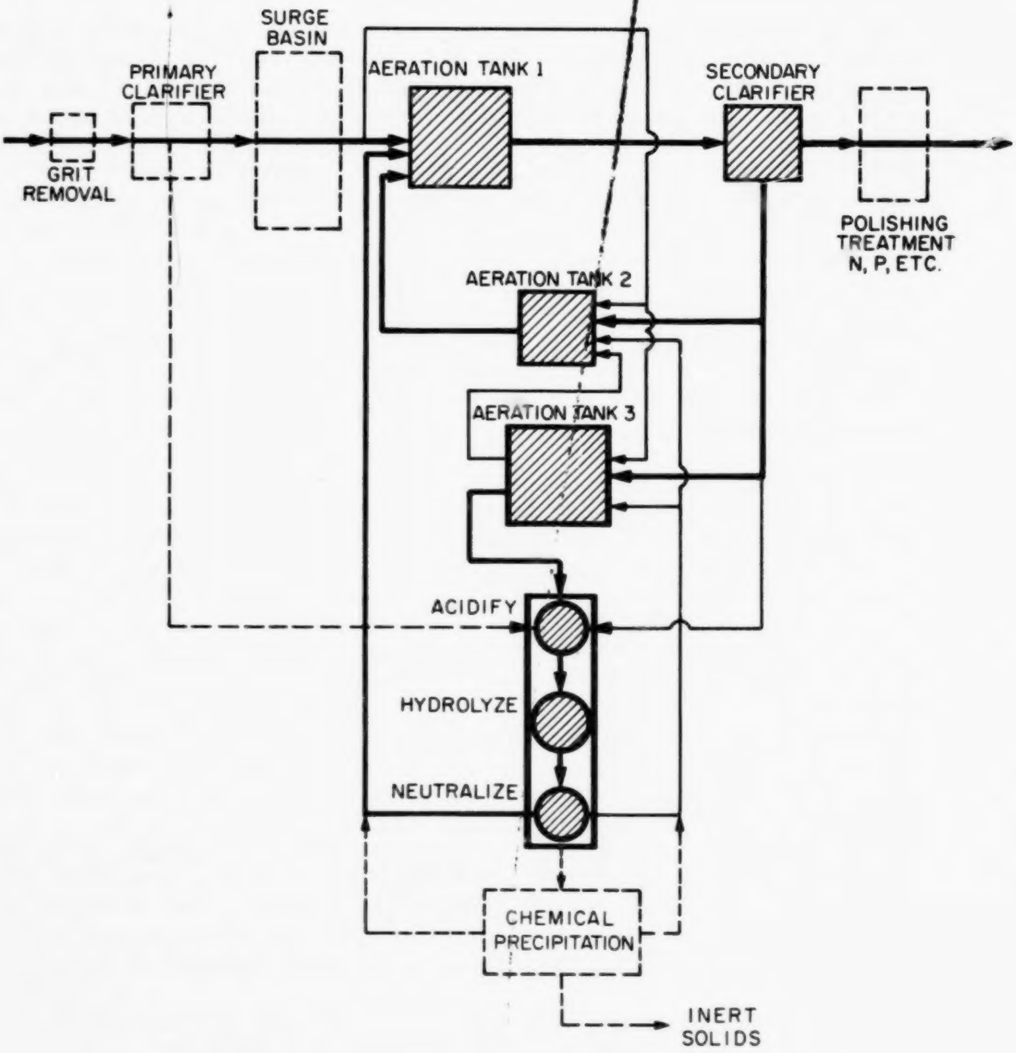
SURGE BASIN
PRIMARY CLARIFIER
AERATION TANK 1
SECONDARY CLARIFIER
GRIT REMOVAL
POLISHING TREATMENT N, P, ETC.
AERATION TANK 2
AERATION TANK 3
ACIDIFY
HYDROLYZE
NEUTRALIZE
CHEMICAL PRECIPITATION
INERT SOLIDS

Figure 35. Possible flow diagram for activated sludge process incorporating suggested modifications for operational control of purification efficiency and including sludge disposal by aerobic autodigestion aided by chemical hydrolysis. This diagram is based on the conceptual principles and research findings presented in this report.

types of organic compounds in the waste are not expected to occur, the surge basin is not needed. The main aerobic reactor (aeration tank 1), the clarifier, and the sludge consistency tank (aerator 2) are in accord with Figure 16, and permit design (and operation) in accordance with the model equations (43 through 47) presented in Chapter V, holding X_R as a system constant. Also, the flow sheet accommodates the principle of total (wet biological) oxidation and employs the hydrolytic assist discussed in Chapter VIII. Thus, disposal of biological solids is effected via the concept of extended aeration, but the substrate removal and the autodigestive phases are separated by design in order that biological solids concentration in aerator 1 and subsequently in the clarifier, can be controlled within operational limits selected by the design. Also, separation of the purification (aerator 1) and "endogenous" phases (aeration tank 3) should enhance control of the latter (i.e., the autodigestive) phase.

Into aeration tank 1 flow the major portion of the influent waste, the recycle sludge X_R from aeration tank 2, and a significant portion of the hyrolysate, and all mixed liquor exits to the clarifier. Into aerator 2, the sludge consistency tank, flow a portion of the clarifier underflow, and three additional inflows. One consists of biological solids from aerator 3; these may be bled in continuously, or as needed to help maintain constant X_R. Both of the remaining inflows consist essentially of substrates (some of the waste inflowing to aerator 1 and some of the hydrolysate). These bleeder lines of inflowing carbon source function essentially as sludge "fresheners," and help pre-condition the biological solids for reentry to aeration tank 1. The sludge has been existing outside of the growth environment of aeration tank 1 for some time, and addition of some of the substrates with which it will shortly be contacted should help to reduce the readjustment or lag period in aerator 1. A lag period at high solids concentration can force oxidative assimilation and, for wastes which contain ample amounts of nitrogen, it is best to curtail oxidative assimilation because this process conserves nitrogen and could possibly increase nitrogen leakage in the effluent.

Aeration tank 3 receives the excess sludge from the clarifier. This tank can be looked upon as a modified aerobic digester which receives, in addition to biological solids, some of the incoming waste and some hydrolysate. The dual function of the aeration tank of an extended aeration plant is here accomplished in two separate tanks, substrate removal in aerator 1 and autodigestion of the excess solids in aeration tank 3. The major portion of the outflow from aeration tank 3 is cycled through the hydrolysis unit, and a small portion may be recycled through aeration tank 2 to aeration tank 1.

The hydrolysis unit receives mixed liquor from aerator 3. Also, provision can be made to pump some clarifier underflow directly to the hydrolysis unit. For wastes containing settleable organic solids, the primary sludge can be channeled to the hydrolysis unit, where the hydrolyzable portion can be liquified and recycled to the treatment plant, i.e., to aeration tank 1, 2, or 3. For waste containing non-hydrolyzable particulate organic matter or suspended inorganic matter, effluent from the hydrolysis unit can be subjected to chemical treatment (shown in the lower portion of Figure 35) to precipitate this material prior to recycling hydrolysate to the treatment plant. These precipitated solids, i.e., non-hydrolyzable particulate organic matter and suspended inorganic solids are labeled "inert solids." They are not inert solids produced in the treatment process, i.e., they are not produced from the biochemically O_2-demanding organic matter in the waste (ΔCOD), but are materials which were initially present as biologically inert organic or inorganic matter. It can be expected that municipal sewage and some industrial wastes would contain some such materials and the chemical treatment unit could prove useful in effecting separation of these "inerts" while the chemical hydrolysis-assisted aerobic digestion is used to provide "ultimate" disposal of the biological solids. The type of flow diagram shown in Figure 35 is, to be sure, a radical departure from current picturizations of activated sludge plants. This flow sheet permits incorporation of new principles and there are various modifications of the flow sheet which could easily be devised.

Figure 36 shows one such modification which uses essentially the same core unit processes in a somewhat different mode of operation. In the flow diagram of Figure 35, the hydrolysis unit was set up for continuous operation. Alternatively (Figure 36), aeration tank 3 and the hydrolysis unit could be operated essentially as a separate extended aeration process which receives a portion of the waste flow, diverted from the inflow to aeration tank 1, and all of the excess sludge from the clarifier. The primary difference between the two flow sheets is the inclusion of a settling compartment in aerator 3 in Figure 36. In this case, the hydrolytic unit could be used intermittently or continuously for control of solids level in aerator 3, as discussed in Chapter VIII (see Figure 34). In addition, aerator 3 could be used to accept increases in flow, thus helping to buffer aerator 1 against hydraulic shocks (changes in F). In both flow diagrams (Figures 35 and 36), the portion of the plant which includes aerator 1, the clarifier, and aerator 2 is designed to operate according to the kinetic model developed in Chapter V (equations 43-47). It should also be pointed out that recycling of hydrolysate to aerator 1 can be used to conserve nitrogen for wastes which are low in nitrogen.

While these flow schemes may seem innovative or imaginative, the conceptual scheme does not require a "far stretch" of the engineering imagination. All new conceptual principles require scrutiny and question by the field and considerable developmental work before being put into general practice. The extrapolation of research findings represented by the flow diagrams in Figures 35 and 36 is presented in order that its pros and cons can be considered by the whole of the profession.

In closing, let us re-emphasize what was said at the beginning: This report is presented as an advancement of conceptual principles, surely

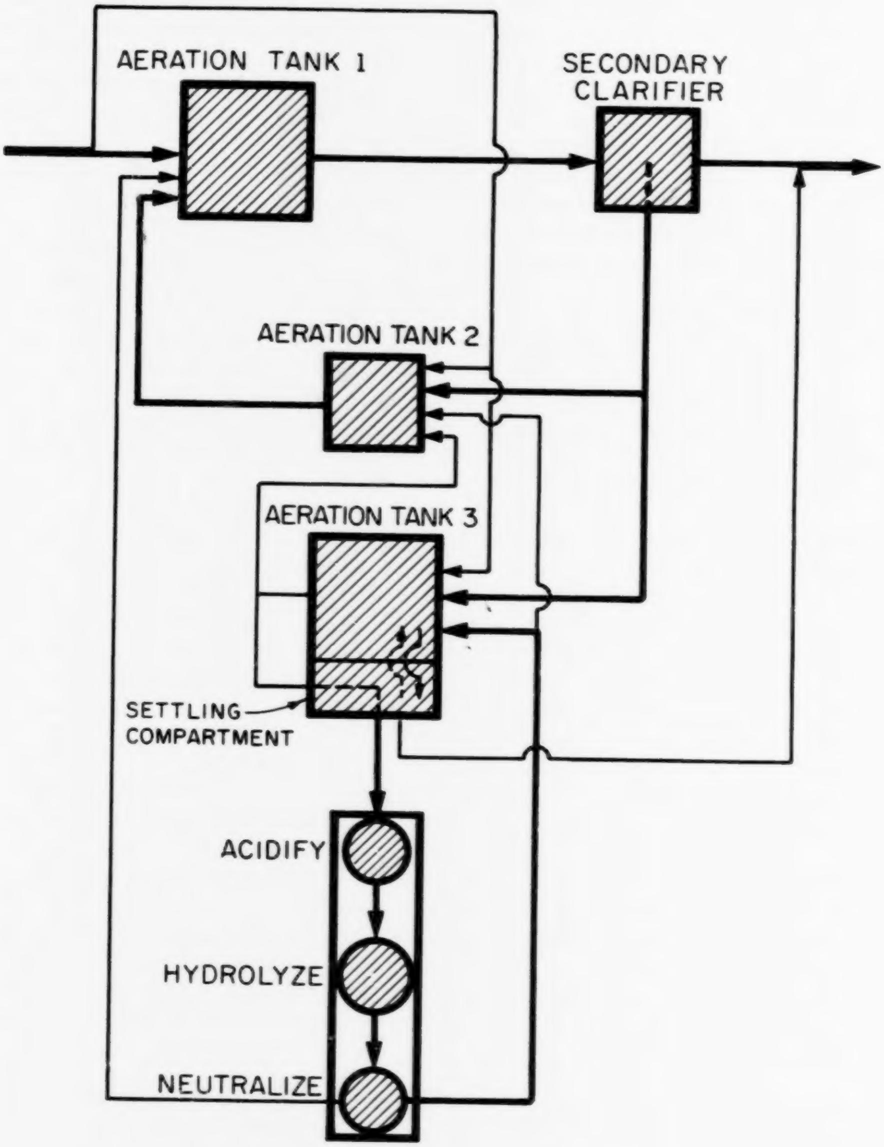
AERATION TANK 1
SECONDARY CLARIFIER
AERATION TANK 2
AERATION TANK 3
SETTLING COMPARTMENT
ACIDIFY
HYDROLYZE
NEUTRALIZE

Figure 36. One of several possible modifications of the flow diagram shown in Figure 35. See text for details.

not an attempt to write finis to principles applicable to biological treatment. The essence of engineering is an ordered and justifiably cautious change based upon definitive scientific fact. We have documented the concepts with investigational findings, and extrapolated the concepts to possible process modifications for further scrutiny by the field. Our aim has been to improve and, if we can hope to write finis to anything at all, it is to the "bio-stasis" and dogmatized concepts with which engineering of the biological process has been traditionally approached.

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PATENTS AND PUBLICATIONS

Although several of the process modifications described herein would seem to constitute patentable art, no patents have been applied for.

Publications completed during the tenure of this research project and supported in part by it include numbers 20, 21, 28, and 32 of the preceding list of references.

GLOSSARY OF TERMS, ABBREVIATIONS, AND SYMBOLS

Definition of Terms	Page introduced
Autotroph - an organism capable of utilizing CO_2 as the sole carbon source for growth	8
Carbon source - a compound which furnishes carbon for synthesis of cellular components	5
Carbonaceous BOD - biochemical oxygen demand due to organic carbon compounds	8
Endogenous phase - metabolism in the absence of exogenous carbon compounds. In a strict sense, for a pure culture this metabolism involves oxidation of its storage materials or other cellular components by an individual cell. In a mixed population, cells may feed on other cells	9
Organotroph - an organism which requires organic compounds as carbon source	8
Oxidative assimilation - utilization of carbon source only for respiration and synthesis of storage products, i.e., without cell replication	100
Photosynthesis - synthesis of organic compounds from CO_2 using light as a source of energy	15
Substrate - any compound which is acted upon by an enzyme; therefore, a compound which is used by the cell for energy and/or synthesis	1
ΔCOD - a measure of the amount of biologically available organic matter in a waste sample	6

Abbreviations and Symbols

A	A collective constant replacing the term $(1 + \alpha - \alpha c)$	58
BOD	Biochemical oxygen demand	5
BOD_5	Biochemical oxygen demand which has been expressed in five days, i.e., y_5	11

Abbreviations and Symbols		Page introduced
c	Sludge recycle concentration factor, equal to the ratio between the recycle solids concentration, X_R, and the biological solids concentration in the reactor, X	56
C_o	Initial concentration	47
C_t	Concentration at any time, t	47
COD	Chemical oxygen demand; measure of total chemically oxidizable organic matter	6
COD_e	Effluent COD exiting the biological reactor system, or COD remaining at the end of a batch experiment	10
COD_{fs}	The COD of biological solids at the end of an experiment	19
COD_i	COD at the beginning of a batch experiment, or influent COD to aerator	10
COD_{is}	COD of the biological solids at the beginning of an experiment	19
COD_w	COD of the waste, not including biological solids if present	19
D	Dilution rate. Ratio of the rate of inflow, F, and the volume of liquor in the aeration tank, V. It is equal to the reciprocal of the mean residence time in a completely mixed reactor	46
D_r	The dilution rate for the reactor alone in systems employing sludge recycle. It is always larger than the dilution rate for the entire system, i.e., $D_r = D(1 + \alpha)$, α being the ratio between the hydraulic rate of recycle flow and the flow of incoming waste, F	56
DO	Dissolved oxygen concentration	5
e	Base of Naperian logarithms	26
F	Rate of flow of incoming substrate or waste water	45

Abbreviations and Symbols		Page introduced
K_s	A biological "constant" used in the hyperbolic expression relating specific growth rate to substrate concentration. It is known as the saturation constant. It is numerically equal to the substrate concentration at which the specific growth rate is equal to ½ the maximum specific growth rate for the system	29
L_o	Ultimate BOD	8
S	Substrate concentration	6
$\bar{S}$	Steady state concentration of substrate in a completely mixed continuous flow reactor	52
S_o	Initial concentration of substrate in batch systems	29
S_i	Concentration of substrate in the inflowing feed in continuous flow operation	45
t	Time	47
$\bar{t}$	Mean residence time in a completely mixed continuous flow reactor, V/F	46
t_d	Doubling time, time required for one doubling in weight of the bio-mass growing in an exponential phase	28
V	Volume of reaction fluid under aeration	45
X	Biological solids concentration, weight per volume	6
$\bar{X}$	Steady state biological solids concentration in a completely mixed continuous flow reactor	52
X_o	Initial biological solids concentration in a batch reactor	26
X_e	Excess biological solids produced in a process (sludge wasted)	56
X_R	Biological solids concentration in the recycle solids flow to an aerobic reactor in a continuous flow system	56

Abbreviations and Symbols		Page introduced
X_t	Biological solids concentration at time t	26
y	BOD expressed, i.e., accumulated oxygen uptake	6
Y	Sludge or cell yield; mg biological solids produced per mg organic substrate used	21
α	Recycle flow ratio in a continuous flow biological reactor system employing cell recycle. It is the ratio between the rate of flow of recycle solids and the rate of flow of incoming waste water	56
Δ	Change in value of a parameter, or difference between initial and final values	6
μ	Specific growth rate in an exponential phase of growth	26
μ_{max}	The maximum specific growth rate for a system in exponential growth	29

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Biological Principles for Design and Operation of the Activated Sludge Process

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27 *Abstract*

Generalized concepts of BOD exertion, the use of ΔCOD as a design and operational tool, the stoichiometry and mass balance concepts of treatment, and kinetic equations for microbial growth are presented. Design models are discussed, and a model for completely mixed reactors holding recycle solids, X_R, constant is recommended. Some guidelines for accommodation of various types of shock loadings are included. Concepts of oxidative assimilation and the multiple effects of solids concentration, nitrogen concentration, and detention time are related; a new activated sludge process (continuous oxidative assimilation) for nitrogen-deficient wastes is presented. Data supporting the concept of total oxidation are presented, and a modification of the extended aeration process incorporating chemical hydrolysis of portions of the sludge is recommended. In the final chapter, some possible flow diagrams for complete aerobic treatment (purification and sludge disposal) of metabolizable organic wastes are presented.

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